PB81-155426

Urban Stormwater and Combined Sewer Overflow Impact on Receiving Water Bodies, Proceedings of the National Conference Held at Orlando, Florida on November 26-28, 1979

University of Central Florida, Orlando

Prepared for

Municipal Environmental Research Lab.
Edison, MI

Dec 80

U.S. Department of Commerce
National Technical Information Service
NTIS

TECHNICAL REPORT DATA

(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA-600/9- 80-056	2.	3. RECIPIENT'S ACCESSION NO. PB81 155426
4. TITLE AND SUBTITLE URBAN STORMWATER AND COMBINED SEWER OVERFLOW IMPACT ON RECEIVING WATER BODIES - Proceedings of the National Conference, Orlando, Florida November 26-28, 1979		5. REPORT DATE December 1980 6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) Yousef A. Yousef, Martin P. Wanielista, Waldron M. McLellon and James S. Taylor - Editors		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS University of Central Florida Orlando, Florida 32816		10. PROGRAM ELEMENT NO. C35B1C 11. /GRANT NO. R-806715
12. SPONSORING AGENCY NAME AND ADDRESS Municipal Environmental Research Laboratory - Cinn., OH Office of Research and Development U.S. Environmental Protection Agency Cincinnati, Ohio 45268		13. TYPE OF REPORT AND PERIOD COVERED Proceedings Nov. 26-28, 1979 14. SPONSORING AGENCY CODE EPA-600/14

15. SUPPLEMENTARY NOTES

Project Officer: Robert Turkeltaub, Storm and Combined Sewer Section, Municipal Environmental Research Laboratory, Cinn., OH, 45268 - FTS 340-6679, (201) 321-6679

16. ABSTRACT

The conference provided a forum for researchers, practioners and others to receive an update on the state-of-the-art and to learn about research findings dealing with stormwater impact. It also served to stimulate dialogue among those who are interested in stormwater effects and control, regarding the implication and applications of current research results, particularly from those projects supported by the U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory's Storm and Combined Sewer Program.

The main topical areas considered included: Combined sewer overflow costs vs. benefits; impacts on lakes, rivers and estuaries; ecological response to stormwater and methodologies for stormwater impact assessment; and stormwater management through the use of receiving water quality models for planning and abatement methodology.

These proceedings contain the contributions from the scheduled speakers and an edited transcription of the taped workshop conducted on practical applications of research findings and future research needs.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Water pollution, Water quality, Combined sewer, Meetings, Impact, Storm sewers, Management, Cost-effectiveness	Urban runoff, Receiving water, Impacts, Control methodologies, Toxics, Biological effects, Sediment, Models, Dissolved oxygen, Pathogens	13B

18. DISTRIBUTION STATEMENT RELEASE TO PUBLIC	19. SECURITY CLASS *(This Report)* UNCLASSIFIED	21. NO. OF PAGES 672
	20. SECURITY CLASS *(This page)* UNCLASSIFIED	22. PRICE

EPA Form 2220-1 (Rev. 4-77)

☆ U.S. GOVERNMENT PRINTING OFFICE: 1980—757 064/0195

United States
Environmental Protection
Agency

Municipal Environmental Research
Laboratory
Cincinnati OH 45268

EPA-600/9-80-056
December 1980

PB81-155426

Research and Development

EPA

Urban Stormwater and Combined Sewer Overflow Impact on Receiving Water Bodies

Proceedings of National Conference
Orlando, FL,
Nov. 26-28, 1979

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA 22161

RESEARCH REPORTING SERIES

Research reports of the Office of Research and Development, U.S. Environmental Protection Agency, have been grouped into nine series. These nine broad categories were established to facilitate further development and application of environmental technology. Elimination of traditional grouping was consciously planned to foster technology transfer and a maximum interface in related fields. The nine series are:

1. Environmental Health Effects Research
2. Environmental Protection Technology
3. Ecological Research
4. Environmental Monitoring
5. Socioeconomic Environmental Studies
6. Scientific and Technical Assessment Reports (STAR)
7. Interagency Energy-Environment Research and Development
8. "Special" Reports
9. Miscellaneous Reports

This report has been assigned to the "SPECIAL" REPORTS series. This series is reserved for reports targeted to meet the technical information needs of specific user groups. The series includes problem-oriented reports, research application reports, and executive summary documents. Examples include state-of-the-art analyses, technology assessments, design manuals, user manuals, and reports on the results of major research and development efforts.

This document is available to the public through the National Technical Information Service, Springfield, Virginia 22161.

EPA 600/9-80-056
December 1980

URBAN STORMWATER AND COMBINED SEWER OVERFLOW IMPACT ON RECEIVING WATER BODIES

Proceedings of the National Conference
Orlando, Florida
November 26-28, 1979

Edited by

Yousef A. Yousef
Martin P. Wanielista
Waldron M. McLellon
James S. Taylor
College of Engineering, University of Central Florida
Orlando, Florida 32816

Based on Grant No. R-806715

Project Officer

Robert Turkeltaub
Storm and Combined Sewer Section
Wastewater Research Division
Municipal Environmental Research Laboratory (Cincinnati)
Edison, New Jersey 08817

MUNICIPAL ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CINCINNATI, OHIO 45268

DISCLAIMER

This report has been reviewed by the Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U.S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health, and aesthetic efforts of pollution. This publication is one of the products of that research; a most vital communication link between the researcher and the user community.

The proceedings contain presentations describing both the impact of urban stormwater and combined sewer overflow on receiving waters and methodologies for stormwater impact assessment for stormwater management. Also included is an edited transcription of the taped workshop.

Francis T. Mayo
Director
Municipal Environmental Research
Laboratory

ABSTRACT

The conference on "Urban Stormwater and Combined Sewer Overflow Impact On Receiving Water Bodies" was held November 26-28, 1979 at Orlando, Florida.

The conference provided a forum for researchers, practitioners and others to receive an update on the state-of-the-art and to learn about research findings dealing with stormwater impact. It also served to stimulate dialogue among those who are interested in stormwater effects and control, regarding the implication and applications of current research results, particularly from those projects supported by the Environmental Protection Agency, Municipal Environmental Research Laboratory's Storm and Combined Sewer Program.

The topical areas considered included:

a. Combined sewer overflow control costs vs. benefits.

b. Impacts on lakes, rivers and estuaries.

c. Ecological response to stormwater and methodologies for stormwater impact assessment.

d. Stormwater management through the use of receiving water quality models for planning and abatement methodology.

The proceedings contained herein include the contributions from the scheduled speakers and an edited transcription of the taped workshop conducted on practical applications of research findings and future research needs.

CONTENTS

Disclaimer ii
Foreword iii
Abstract iv
Acknowledgments viii

1. CONFERENCE SUMMARY 1

2. FIRST SESSION - GENERAL OVERVIEW 7

Urban Runoff Receiving Water Impacts - Program Overview and Research Needs
Richard Field and Robert Turkeltaub 8

3. SECOND SESSION - IMPACTS ON LAKES 31

An Assessment of the Impact of Urban Drainage on Eutrophication-Related Water Quality in Urban Lakes
G. Fred Lee and R. Anne Jones 32

The Effect of Urban Stormwater Runoff on the Water Quality of Lake Jackson, Florida
Christian Byrne, C.R. Donahue and W.C. Burnett 58

A Comparison of Storm-Related Material Loadings to Two Glacial Lakes from Urban, Wetland, and Agricultural Sources
Robert P. Glandon, Frederic C. Payne, Clarence D. McNabb, and Ted R. Batterson 72

4. THIRD SESSION - IMPACTS ON LAKES AND RIVERS 89

Impacts of Stormwater on a Florida Lake Ecosystem: Effects on Water Quality and Biota
Eldon C. Blancher, II 90

The Distribution of Sediments and Particulate Contaminants from Combined Sewer and Storm Drain Overflows in Seattle's Nearshore Waters

R.D. Tomlinson, B.N. Bebee, S. Lazoff, R.G. Swartz and D.E. Spyridakis 115

The Ecological Effects of Urban Runoff on Stream Communities
Donald B. Porcella 147

5. FOURTH SESSION - IMPACTS ON RIVERS 167

Analysis of Receiving Stream Impacts on the Milwaukee River
Thomas L. Meinholz 168

Urban Stormwater Impacts on Receiving Streams in North Carolina
E. Ryland Brown and Ross S. Green 197

Dissolved Oxygen Impact from Urban Stormwater Runoff
Thomas M. Keefer, Robert K. Simons and Raul S. McQuivey . . . 223

The Impact of Combined Sewer Overflows on the Dissolved Oxygen Concentration of a Small Stream
Thorkild Hvitved-Jacobsen 245

6. FIFTH SESSION - IMPACTS ON ESTUARIES 260

Impacts of Intermittent Discharges on Receiving Waters
John L. Mancini 261

The Response of Great Lake Estuaries to Stormwater Runoff
John R. Adams and Stephen M. Yaksich 281

Water Quality and Urban Runoff in Selected Canal Communities Along Texas Coast
Allen Messenger and Tom D. Reynolds 301

7. SIXTH SESSION - ECOLOGICAL RESPONSE TO STORMWATER 323

The Response of Infaunal Communities to Seasonal Variations in Wastewater Discharge
Ronald M. Thom 324

Productivity Responses of Lake Eola Water to Urban Runoff
Harvey H. Harper, III, Yousef A. Yousef, and Martin P. Wanielista 341

Water Quality and Biological Effects of Urban Runoff on Coyote Creek
Robert Pitt and Martin Bozeman 371

8. SEVENTH SESSION - METHODOLOGIES FOR STORMWATER IMPACTS ASSESSMENT . 406

Nationwide Assessment of Urban Stormwater Impacts on Receiving Water Bodies
James P. Heaney, Wayne C. Huber and Melvin E. Lehman 407

Statistics of Advective Dispersive System Response to Runoff
Dominic M. DiToro 437

Continuous Receiving Water Quality Modeling for Urban Stormwater Management
Miguel A. Medina, Jr.. 466

Potential of Stormwater Impacts Based on Comparative Analysis of Wet and Dry Weather Pollutant Loading
Douglas C. Ammon and Richard Field 502

9. EIGHTH SESSION - STORMWATER MANAGEMENT 523

The Use of Receiving Water Quality Models in Urban Runoff Pollution Abatement: Application to Marginal Benefit: Marginal Cost Analysis
Cornelius B. Murphy, Jr., Gregory J. Weller and Dwight A. MacArthur 524

Methodology For Evaluating the Impact and Abatement of Combined Sewer Overflows A Case Study of Onondaga Lake New York
Peter E. Moffa, John C. Byron, Steven D. Freedom and John M. Karanik 557

A Water Quality Planning Methodology for Urban Areas
Franklin W. (Skip) Ellis and Ronald L. Wycoff 590

10. WORKSHOP ON PRACTICAL APPLICATIONS AND RESEARCH NEEDS FOR RECEIVING WATER RESPONSES TO URBAN STORMWATER

Session I 618

Session II 631

Session I - Summary
James P. Heaney 650

Session II - Summary
John L. Mancini 651

APPENDIX

List of Conference Participants 653

ACKNOWLEDGMENTS

We wish to acknowledge with gratitude the assistance of the many individuals who helped make this conference possible. The financial and technical support given by the United States Environmental Protection Agency, Municipal Environmental Research Laboratory, Storm and Combined Sewer Program and the American Society of Civil Engineers, Florida Section is very much appreciated.

Special acknowledgments are extended to Mr. Richard Field and Mr. Robert Turkeltaub, EPA, Storm and Combined Sewer Program, Edison, New Jersey for their invaluable contribution and assistance in the review process and finalizing the program.

Appreciation is also extended to all speakers for their desire to share their knowledge and to all participants who made this conference a success.

Special recognition must go to personnel from the College of Extended Studies and the College of Engineering at the University of Central Florida who helped arrange for the conference, edit the proceedings and type the manuscript.

Yousef A. Yousef, Ph.D., P.E.

"URBAN STORMWATER AND COMBINED SEWER OVERFLOW IMPACT ON RECEIVING WATER BODIES"

CONFERENCE SUMMARY

INTRODUCTION

Urban stormwater and combined sewer overflow (CSO) have been identified as significant pollutant sources. Runoff from urban areas is usually contaminated with organic matter, sediments, nutrients, heavy metals, pesticides, bacteria and other toxic substances. These pollutants will eventually enter the adjacent environment in significant quantities depending on many factors such as antecedent dry period, land use of the drainage basin, social and economic status, degree of urbanization and volume and type of traffic, industry and air pollution fallout. The magnitude of these pollutant loadings has led to a definite national need to study receiving water impacts.

Wet weather conditions and associated runoff may greatly affect the quality of receiving waters due to shock effects and long-term buildup. The extent to which the aquatic environment will be affected and courses of remedial action can only be determined by experience and/or experimentation. Without experience and knowledge, logical decision must depend on experimentation. Decision makers throughout the country are looking for information on receiving water impacts in order to preserve the integrity of their streams and to abide by the goals set forth in PL 92-500 and its amendments.

Conferences provide a forum for effective and rapid dissemination of available information. Also researchers and planners in all levels of Federal, State and local government and the private sector are willing to present state-of-the-art information and to engage in helpful discussions. This fact was realized by the Storm and Combined Sewer Program, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency who encouraged and supported efforts by the Department of Civil Engineering and Environmental Sciences, College of Engineering, University of Central Florida to host a national conference on "Urban Stormwater and Combined Sewer Overflow Impact on Receiving Water Bodies," November 26-28, 1979 at Orlando, Florida. The conference was also sponsored by the American Society of Civil Engineers, Florida Section. This conference drew national and international attention judging from the list of participants and requests for the proceedings.

CONFERENCE FORMAT

The conference was divided into eight different sessions for formal presentations on November 26-27, 1979 and was followed by a workshop on

November 28. The opening session set the theme for the conference. Myron Tiemens, EPA, Washington, D.C., started the technical presentations with a presentation on "Combined Sewer Overflow Control: Costs vs. Benefits" (no formal paper). His presentation was followed by general remarks (no formal paper) from Dennis Athayde, EPA, Washington, D.C.; then Robert Turkeltaub and Richard Field, EPA, Edison, New Jersey followed with a presentation on "Urban Runoff Receiving Water Impacts - Program Overview and Research Needs." The other sessions covered studies and analysis demonstrating methodologies for stormwater impacts assessment and stormwater management. The workshop was divided into two sessions to discuss practical applications and research needs for receiving water responses to urban stormwater.

The conference was attended by approximately 180 persons in addition to 40 speakers and moderators. The magnitude and complexity of the problem was evident from discussions that took place by chemists, biologists, engineers, economists, management and planning personnel and others who participated in the conference. The sessions were very well attended and good interaction between speakers and attendees was developed. This voluminous proceedings represents the contribution from participating speakers and edited copy of the taped workshop.

SUMMARY OF PRESENTATIONS

During the first general session, Myron Tiemans, EPA, Washington, D.C. indicated that secondary treatment requirements will be met before a CSO remedial program is initiated and funding will be allocated for pollution reduction only. Also, planning money for CSO will be accelerated by construction grants after they are provided better cost estimates, better enforcement policy and better plan review. He was followed by Dennis Athayde, EPA, Washington, D.C., who stated that the U.S. EPA's National Urban Runoff Program (NURP) will be determining effects of urban runoff on receiving waters. The first session was ended by Robert Turkeltaub, EPA, Edison, New Jersey, who presented a comprehensive paper on the status of the U.S. EPA's Storm and Combined Sewer Program receiving water impacts efforts and future research needs.

The second technical session started with G. Fred Lee, Colorado State University, Fort Collins, Colorado, who presented an approach for assessing the water quality significance of chemical contaminants in urban lakes. Dr. Lee stated that the OECD (Organization for Economic Cooperation and Development) results should be used to describe the nutrient - load response relationship for urban lakes and most importantly, should be used to estimate the magnitude of the water quality improvement resulting from reduction of nutrient loads to phosphorus limited lakes. Also, an environmental hazard assessment based on aquatic toxicology and environmental chemistry - fate information should be conducted to define the magnitude of the impact of toxicants on beneficial uses of receiving water bodies. This was followed by Christian J. Byrne, Florida State University, Tallahassee, Florida, who investigated the petroleum hydrocarbon content in urban stormwater and initiated studies to model the hydrocarbon budget of Lake Jackson, Florida. The second session was concluded by Robert P. Glandon, Michigan State

University, East Lansing, Michigan, who showed a comparison of storm-related phosphorus and nitrogen loading to two glacial lakes in Lake Michigan drainage basin, from urban, wetland and agricultural sources.

The third session started with Eldon C. Blancher II, Marine Environmental Sciences Consortium, Dauphin Island, Alabama, who presented data indicating a direct impact of stormwater runoff on water quality and subsequent changes in biota of Lake Conway ecosystem located in Orange County, Florida based on Secchi disk transparency, chlorophyll a and total zooplankton numbers. Richard D. Tomlinson, Municipality of Metropolitan Seattle, Seattle, Washington, discussed the distribution of sediments and particulate contaminants from combined sewer and storm drain overflows in Lake Washington. This session was concluded by Donald B. Porcella, Tetra Tech, Inc., Lafayette, California, who presented a literature review of the ecological effects of urban runoff on streams. He discussed the concept that large scale variations and instabilities of stream pollutant input and concentration would result in greater impact to stream communities than steady inputs.

Thomas L. Meinholz, Ecolsciences, Inc., Milwaukee, Wisconsin, started session four and presented an analysis of receiving stream impacts on the Milwaukee River. He reported tremendous DO sags in the lower reaches of the river following runoff events due to scouring of the bottom sediments by submerged combined sewer outfalls. He was followed by E. Ryland Brown, North Carolina Department of Natural Resources, Raleigh, North Carolina, who predicted that, under present conditions, almost all urban streams in North Carolina will be unable to meet the 1983 water quality goals. Thomas N. Keefer, The Sutron Corporation, Arlington, Virginia, examined records from 104 water quality monitoring sites throughout the country. He concluded that the stream flow increases and the dissolved oxygen diurnal cycle disappears following a storm event. The minimum DO drops from 1 to 1.5 mg/l below the minimum values observed during steady flows and remains constant there for periods ranging from one to five days. Thorkild Hvitved-Jacobsen from Denmark talked about immediate and delayed effects on the DO concentration in a stream following a storm event.

The fifth session covered impacts of storm and combined sewer overflow on estuaries. John Mancini, Manhattan College, Bronx, New York, presented methodologies to illustrate impacts of intermittent CSO discharges on bathing beach areas of New York City and discussed the fate and effect of intermittent discharges of toxics. Then John R. Adams, Army Corps of Engineers, Buffalo, New York, discussed the sources of pollution in the estuaries reach of the Maunee River. He outlined the problems and complexities of chemical measurements of water quality in estuarine systems of the Great Lakes during varying hydrologic conditions. This session was concluded by Allen L. Messenger, Austin, Texas, who presented data to demonstrate deterioration in water quality of canals in the Galveston Bay Area resulting from stormwater runoff and possible resuspension of bottom sediments.

The sixth session started with Ronald M. Thom, University of Washington, Seattle, Washington, discussing the response of infaunal communities to seasonal variations in CSO discharges. He indicated that the communities at the sites nearest the overflow appeared to be primarily affected by intense scouring and deposition, whereas those communities further away appeared to respond largely to input of organic debris. He was followed by Harvey H. Harper, University of Central Florida, Orlando, Florida, who presented results from algal bioassay on impact of runoff on Lake Eola water. He demonstrated the impact of nutrients and toxic substances released to Lake Eola in urban runoff water. The session was concluded by Robert Pitt, Woodward Clyde Consultants, San Francisco, California, who presented his study to demonstrate biological effects of urban storm runoff on Coyote Creek, San Jose. Fish, benthic macroinvertebrate and attached algae were used as indicators and concluded that the urbanized portion of the creek is significantly degraded by urban stormwater.

The seventh session on methodologies for stormwater impact assessment was started by James P. Heaney, University of Florida, Gainesville, Florida, who indicated that well-documented cases of receiving water impacts are scarce. His nationwide assessment will be used as a basis for devising simple criteria for analyzing an urban area to determine whether a potential impact does or would occur. He was followed by Dominic M. DiToro, Manhattan College, New York, who presented methods to evaluate the statistical behavior of receiving water concentrations that result from runoff events. These methods are specially useful for screening many alternative treatment schemes that abate runoff mass discharge. He was followed by Miguel A. Medina, Jr., Duke University, Durham, North Carolina, who presented Level III - Receiving Water Quality Model. This model has been developed to permit preliminary planning and screening of area-wide urban wastewater treatment alternatives, in terms of frequency of water quality violations and more traditional approaches such as dissolved oxygen profiles. This session was concluded by Douglas C. Ammon, EPA, Edison, New Jersey, who summarized much of the stormwater characterization data for trace contaminants and several conventional parameters. He also identified the significance of each contaminant.

Session eight discussed stormwater management tools. Gregory J. Welter, O'Brien and Gere Engineers, Inc., New York, New York, discussed the use of receiving water quality models in the analysis of the Syracuse, New York; Rochester, New York and the District of Columbia combined sewer facilities planning activities using the marginal cost - marginal benefit approach. He was followed by Peter E. Moffa, Stearns and Wheler, Cazenovia, New York, who presented a general methodology for the evaluation of the impact and abatement of CSO on receiving waters. This methodology was developed from experience with Onondaga Lake in Central New York. A 27-segment, three-dimensional, dynamic water quality model of the lake was developed with capability of predicting enteric bacteria, dissolved oxygen, nutrients and toxic materials as a function of pollutant load. The model made it possible to predict DO deficit and fecal coliform content following treatment alternatives for CSO. The technical sessions were concluded by Franklin W. (Skip) Ellis, CH_2M Hill, Reston, Virginia, who presented a two-phase approach to water quality planning for urban areas. Phase I involves the use of the continuous stormwater pollution simulation system (CSPSS) for economic optimization for various levels of pollution abatement. Phase II will produce a description of the optimal mix of control alternatives, the total plan costs and the receiving water quality response.

SUMMARY OF WORKSHOP

Upon completion of the technical presentations, arrangements were made for follow-up discussions and interactions between speakers and attendees during a two-session workshop held November 28, 1979. Each session lasted for one and one-half hours. It started by opening statements from five panelists for five minutes each followed by discussion from the floor. A summary of important comments presented during the workshop follows:

1. There are very limited cases where evidence of stormwater and CSO impacts on receiving water bodies is adequately documented. Also, there is no quality assurance in data collected. A simple and reliable quality control methodology needs to be developed.

2. Chemical standards and pollutional loadings alone are not adequate criteria to determine impacts. Biological indicators need to be used and biological sampling must represent various levels of food chain.

3. Bioassay methodologies need to be improved and expanded in order to be able to predict biological response as related to changes in chemical constituents.

4. Fate and effects of toxic substances and heavy metals released to receiving streams and their relationship to public health problems need to be investigated.

5. Water quality problems should be defined as those which interfere with stated beneficial water usage. The magnitude and frequency of the problem should be investigated and supplemented with a realistic cost benefit analysis. Therefore, simplified methods of assessing benefits need to be developed.

6. Frequency-duration-response curves for stormwater and CSO release to receiving water need to be developed and measured in terms of impairment of beneficial use.

7. Modeling of stream impacts is a complex process which needs better identification of parameters involved. Fundamental research is needed to define deoxygenation and reaction kinetics, transport and fate of pollutants, sediment transport and its impact on benthic uptake and other geophysical and hydrodynamic processes.

8. Simplification of existing models is desired and guidelines for better matching the model to data base are needed. Models based on statistical analysis could be useful and a process for model calibration and verification is needed.

9. Socio-political constraints should be considered in assessing receiving water impacts.

In conclusion, the conference was well attended, good presentations were made and the response was encouraging. Stormwater impacts will continue to attract more attention and the need for continued investigation will increase. Similar conferences are most beneficial for fast dissemination of available information.

First Session

GENERAL OVERVIEW

Moderator: Carl Brunner
EPA, Cincinnati, Ohio

URBAN RUNOFF RECEIVING WATER IMPACTS: PROGRAM OVERVIEW AND RESEARCH NEEDS

by

Richard Field* and Robert Turkeltaub**
Environmental Protection Agency, Edison, New Jersey

ABSTRACT

Recieving water impacts are a major national concern. We are spending huge sums of money on secondary treatment plants, meanwhile major culprits, stormwater and combined sewer overflows, are still uncontrolled. To attain the goals set forth in PL 92-500 and PL 95-217 of abating pollution and achieving water quality standards in an economical and efficient manner those analyzing, planning and designing controls must have a thorough understanding of the impact of pollutants on receiving waters. Receiving water impacts are the bottom line justification for funding countermeasure campaigns and the passage of abatement legislation. This conference will provide a forum for the attendees to acquire first hand knowledge of the state-of-the-art and to consider ongoing and recently completed research.

Data on the environmental impacts of urban stormwater and combined sewer overflow are being gathered by projects of the Storm and Combined Sewer Program (SCSP) of the Municipal Environmental Research Laboratory (MERL) as a first step in developing control needs and a methodology to quantify pollutant stress and evaluate the impact in relation to receiving water standards and desired uses. This paper will contain a brief history of our ongoing projects. The projects will be briefly described including project objectives and an outline of significant results to date. Also, future Program needs will be discussed and areas in which we anticipate concentrating our efforts will be outlined.

BACKGROUND

Recognition of the significance of stormwater-induced water pollution has been slow. Current interest is largely attributable to the efforts of the USEPA Storm and Combined Sewer Research, Development, and Demonstration Program.

---

*Chief, Storm & Combined Sewer Section, Municipal Environmental Research Laboratory (Cincinnati), U.S. Environmental Protection Agency, Edison, New Jersey 08817

**Staff Engineer, Storm & Combined Sewer Section, Municipal Environmental Research Laboratory (Cincinnati), U.S. Environmental Protection Agency, Edison, New Jersey 08817

Only within the past 20 years, has it been recognized that waters discharged from separate storm sewers contain pollutants. Rain falling on an urban area picks up pollutants from the air, dusty roofs, littered and dirty streets and sidewalks, traffic by-products (gasoline and oil drippings, tire residuals, brake ware, vehicular exhaust), metallic corrosion, hazardous spill materials and chemicals applied for fertilization, control of ice, rodents, insects, and weeds.

Urban Runoff Characterization

Figure 1 illustrates representative strengths of wastewaters. The average five-day biochemical oxygen demand (BOD_5) concentration in combined (domestic and storm) sewer overflow is approximately one-half the raw sanitary sewage BOD_5. However, storm discharges must be considered in terms of their shockloading effect due to their great magnitude. A common rainfall can produce flow rates up to 100 times dry-weather flow. Even separate stormwater is a significant source of pollution, having solids concentrations equal to or greater than untreated sanitary wastewater, and BOD_5's approximately equal to secondary effluent. Bacterial contamination of separate stormwater is two to four orders greater than concentrations considered safe for water contact.

Microbiological studies of storm runoff have shown consistent recoveries of pathogenic organisms (1). This indicates that all types of urban runoff, in general, may be hazardous to health.

Representative Loads

From 40 percent to 80 percent of the total annual organic loading entering receiving waters from a city is caused by sources other than the treatment plant. During a single storm event, 95 percent of the organic load is attributed to wet-weather flow sources. The runoff of toxic pollutants, particularly heavy metals, and petroleum hydrocarbons is also high.

Potential Impacts

Approximately one-half of the stream miles in this country are water quality limited and 30 percent of these stream lengths are polluted to a certain degree with urban runoff. Hence, generally speaking, secondary treatment of dry-weather flow (DWF) may not be sufficient to produce required receiving water quality; and control of runoff pollution becomes an alternative for maintaining stream standards. For example, Figure 2 illustrates that if Durham, North Carolina provided 100 percent treatment of municipal wastewater, the total annual reduction of ultimate BOD and suspended solids would only be 59 percent and 5 percent, respectively. Durham has a separate system. Combined systems offer a greater pollutional impact since additional loads come from domestic wastewaters, dry-weather sediment washout, and more impervious and populated lands.

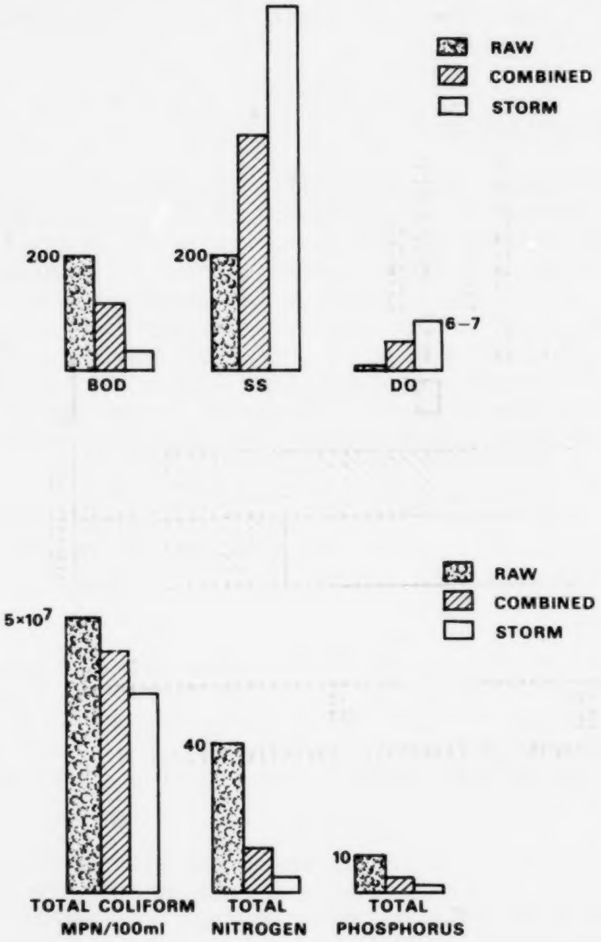


Figure 1. Representative Strength of Wastewaters (Flow Weighted Means in mg/l)

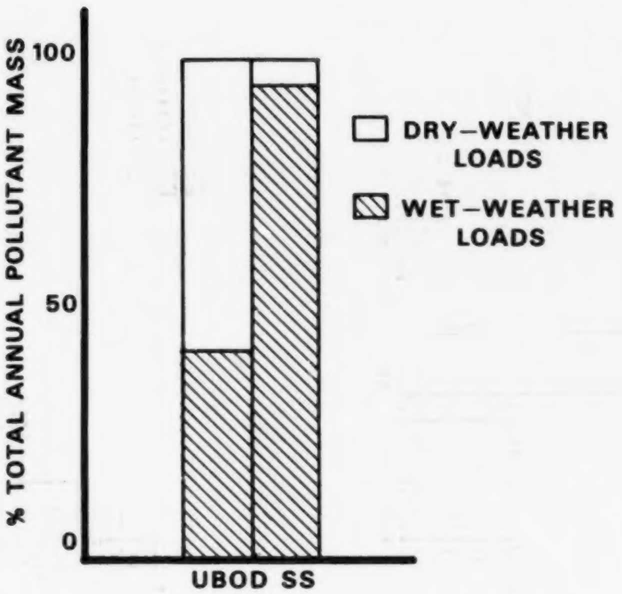


Figure 2. Total Wet-Weather and Dry Weather Pollutant Load - Durham, North Carolina

Now that urban runoff has been identified as a significant pollutant source the emphasis has been refocused from direct analysis of urban runoff to substantiating its impacts on receiving waters and developing source control methodologies based on these impacts. Identified and quantified impacts and their associated impairment of designated beneficial uses of receiving waters are fundamental factors providing justification for funding stormwater and CSO countermeasure campaigns and passing abatement legislation.

For instance, in order for a CSO control project to qualify for Federal funding a careful review must have shown that out of all the alternatives considered, the selected alternative is cost effective and required to protect the beneficial use of the receiving water, even after industrial effluent limitations and secondary treatment for dry-weather flows are achieved (2).

Very little information of this kind can be found; therefore, several years ago the U.S.EPA Storm and Combined Sewer (SCS) Program began a modest effort to fill this data void. Ties between receiving water quality and stormwater discharges must be clearly established and delineated. Quantification of the impairment of beneficial uses and water quality objectives by such discharges is a major goal. Results from earlier SCS Program projects and limited results from others are now available. The basic objective of this paper is to present an overview of the SCS Program's receiving water impacts subprogram. Many of the Program's projects will be described in detail during the course of this conference.

RECEIVING WATER QUALITY IMPACTS

Data on the environmental impact of urban runoff are being gathered as a first step in developing methodology to quantify pollutant stress and evaluate the impact in relation to receiving water standards and desired uses.

Dissolved Oxygen Depletion

The classical problem related to organic pollution of receiving waters is the consumption of instream oxygen by the bacterial breakdown of organic material. The resulting low levels of oxygen will destroy sensitive species of fish and aquatic organisms and may cause anaerobic conditions which produce objectionable end products. Accordingly, the SCS Program's first attempt at evaluating receiving water impacts was a project by the Sutron Corporation (3) wherein data from existing rain gages, flow and water quality monitors in and downstream of urban areas was assessed in an attempt to correlate dissolved oxygen (DO) depletion and urban runoff.

Although sites with specific instances of standards violations were located, no conclusions could be drawn on a national basis. Several other correlation attempts in Ohio and Indiana streams also proved fruitless (4, 5).

Except for one occurrence in the Chattahoochee River, after a July thunderstorm, when the DO reached a low of 1.5 mg/l the Chattahoochee project (6) also failed to ascertain adverse DO impacts despite BOD_5 and other pollutant concentrations in the CSO ranging from 1 to 10 times that in the receiving streams.

The 1.5 mg/l DO level recorded in the Chattahoochee River is an indication of the potential negative impacts from oxygen demanding material in urban storm runoff. At this point although the SCS Program has not had complete success in ascertaining DO impacts from urban runoff the Program should not attempt to write off DO demand as being a wet-weather related impact. Based on annual mass balance determinations urban wet-weather organic loads are the same order of magnitude as dry-weather loads as indicated by Table 1, and 10 times greater during storm periods (7, 8). However, a large fraction of the oxygen consuming material is associated with settleable and separable solids which in storm flow are usually larger and in a more non-biodegradable state than in sanitary effluent therefore the deoxygenation rate tends to be lower and time delayed. The fate of this material is related more to the hydrodynamics of the specific receiving water and to the specific wet-weather oxygen transfer rates rather than to the advective transport and lumped and conventional rate assumptions used in traditional DO analysis.

Table 1. National Annual Urban Wet- and Dry-Weather Flow BOD_5 and COD Load Comparisons*

TYPE	PERCENT OF DEVELOPED AREA	ANNUAL DWF**		ANNUAL WWF**		PERCENT WWF	
		BOD_5 MIL LB.	COD MIL LB.	BOD_5 MIL LB.	COD MIL LB.	BOD_5	COD
COMBINED	14.3	340	910	880	2640	72	74
STORM	38.3	710	1890	440	2500	36	57
UNSEWERED	47.4	310	830	360	2250	54	73
TOTALS	100	1330	3630	1640	7390	55	67

*ASSUMING

	BOD_5 (MG/L)	COD (MG/L)
CSO	100	300
URBAN STORMWATER	20	125
DRY-WEATHER	30	80

**LB = 0.454 KG

Part of the problem in ascertaining DO impacts lies in locating urban areas free from all interferences except urban runoff. And, as Figure 3 indicates, part lies in trying to derive data from sites set up for conventional dry-weather monitoring applied for unsteady state flow regimes caused by storms. Also, storm related changes in deoxygenation rate, stream velocity and reoxygenation rate are variables which can cause the DO sag to occur at remote locations as compared to relatively steady state dry-weather conditions. And still another factor is money. Receiving water studies are expensive and the SCS Program and others can ill afford to spend substantial sums from their limited budgets on one project.

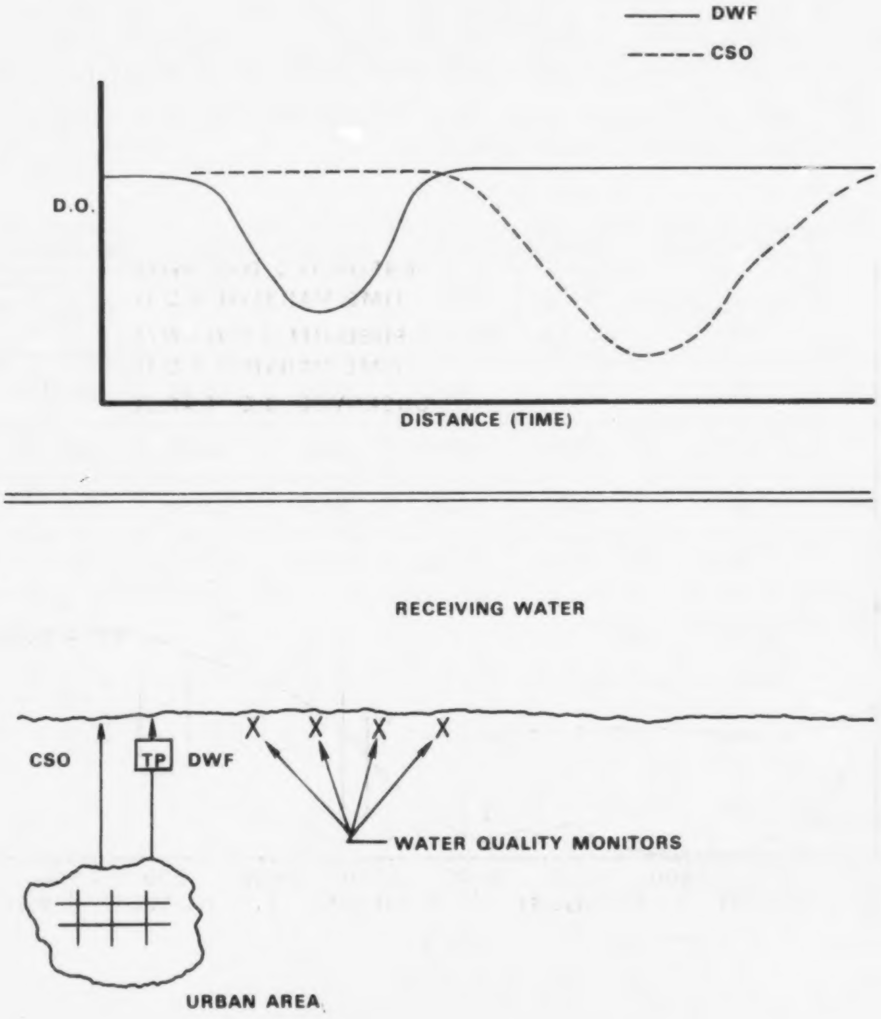


Figure 3. Comparative Locations of DO Sag and Water Quality Monitors

In Milwaukee, these problems were somewhat overcome (9). The City was desirous of providing the necessary receiving water impact analyses for evaluating CSO pollution and control and the SCS Program was able to interface with their 201, step 1 construction grant.

River sampling results provided strong evidence of CSO impacts on intensifying DO sag. The study has recently been completed and it was found that resuspension of the sediments caused rapid DO sags in the Milwaukee River after CSO events. Combined sewer overflows were found to contribute 40 to 50 percent of the annual BOD loading to the sediments and the sediment oxygen demand (SOD) of the disturbed sediments was 100 times greater than SOD rates in situ (10 g O_2/m^2/day vs. 1000 g O_2/m^2/day). Figure 4 illustrates the marked effect that the disturbed sediment has on the DO in the river. The I-beams represent the range of the observed real river data and offers verification of model predictions using disturbed SOD rates.

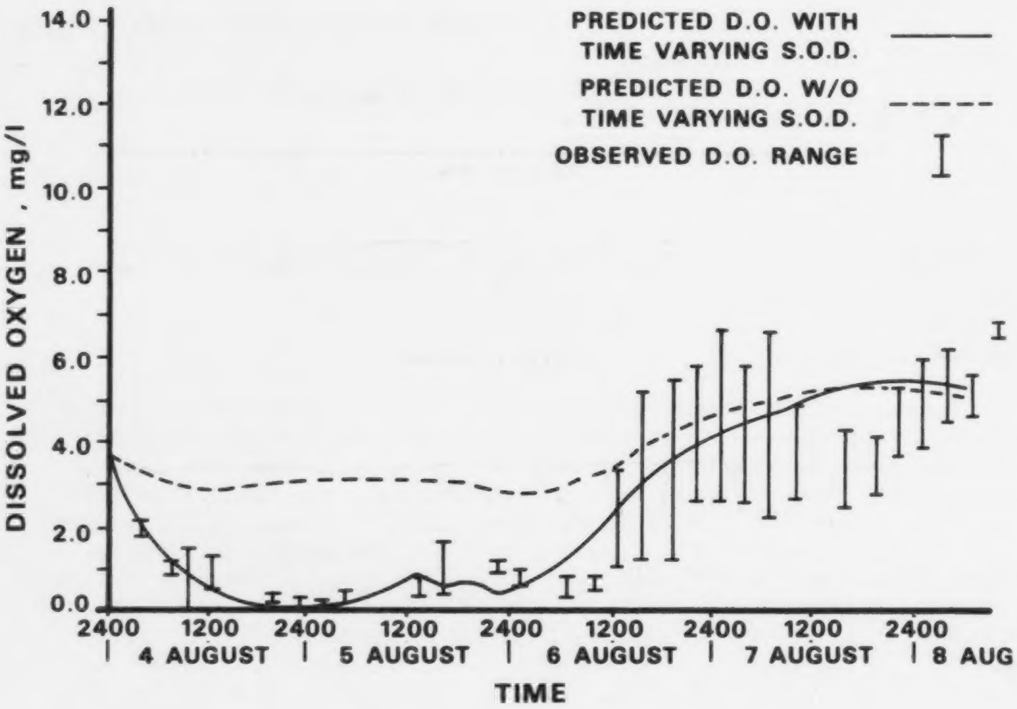


Figure 4. DO Storm Profile - Milwaukee River, Milwaukee, Wisconsin

Based on results identifying sediment as a major problem, the SCS Program undertook a project to track the fate of settleable materials discharged to flowing streams (10). The project surfaced limitations with the current state-of-the-art on transport modeling.

Conventional receiving water quality models exclude sediment transport while existing sediment transport models only consider discrete sediments and not biochemical reactions. Furthermore, the present data base is inadequate to validate these sediment transport models. For them to be useful in wet-weather discharge analysis, it will be necessary to interface the sediment routines with the runoff routines and to build-in the appropriate biochemical reactions. As indicated by Figure 5, the questions boils down to the level of effort and cost-effectiveness in developing a more complex model and data base.

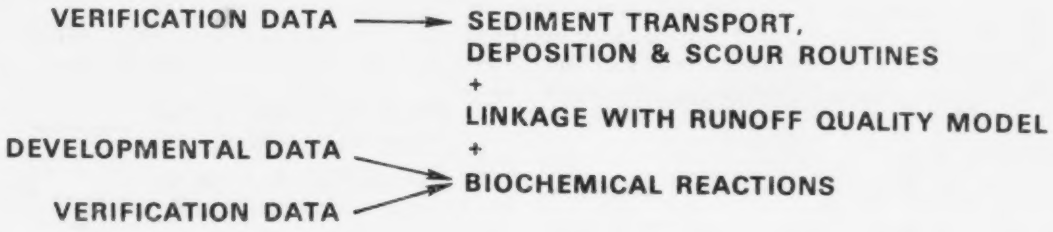


Figure 5. Receiving Water Sediment/Pollutant Impact Analysis

Pathogen Concentrations

Excess concentrations of bacterial indicator organisms in urban runoff will hinder water supply use, recreational use and fishing/shell fishing use of the receiving water. A common bacterial standard for recreational use of water is a total coliform concentration of less than 1000 organisms/ 100 ml and a fecal coliform concentration of less than 200 organisms/100 ml. However, coliform are merely indicating organisms and not direct pathogens. Their use as an indicator of human enteric contamination in storm runoff waters is not valid because some coliform in this flow originates from soils and animal fecal matter inherent in storm runoff. Therefore, the SCS Program supported the Johns Hopkins University study (1) whose conclusions showed that the frequency of occurrence of human pathogenic organisms in storm flow relates to cross contaminations from sanitary sewage. All stormwaters were found to contain disease causing viruses and bacteria because none of these waters are totally separated from sanitary or other contaminating waste sources. As a result some pathogens were even found in the background waters used for control observations.

The aforementioned Milwaukee project (9) found that CSO is the major source of fecal coliforms in the Milwaukee River and that removal of these overflows will nearly eliminate fecal coliform standards violations in the river.

In a study conducted at Syracuse, New York (11) the only water quality parameter which was significantly and adversely affected by CSO into Lake Onondaga was fecal coliform. As Figure 6 illustrates, of the 65 annual CSO producing events, 38 resulted in discharges of significant magnitude to cause violation of the bacterial standards in the contact recreational zone of the lake. A hypothetical one-year, two-hour storm would result in bacterial numbers in excess of the standard of 200 cells/100 ml in all "upper" segments of the lake, and a maximum concentration of greater than 11,000 fecal coliform cells/100 ml in the northern basin where contact recreation is anticipated. A storm of this magnitude would cause violations of coliform standards for a period in excess of three days. However, it is probably not necessary to treat a one year frequency storm. It can be seen from Figure 6 that treatment provided for a two month frequency storm will eliminate 82 percent of the coliform violations. The cost of providing additional treatment for the larger storms at the so-called knee of the curve is disproportionately large compared to the rapidly decreasing benefits achieved much beyond this point on the curve. It should be emphasized that the concept of disproportionate returns is a decision-maker's tool and that the bottom line lies with the desired frequency of water usage at the local level.

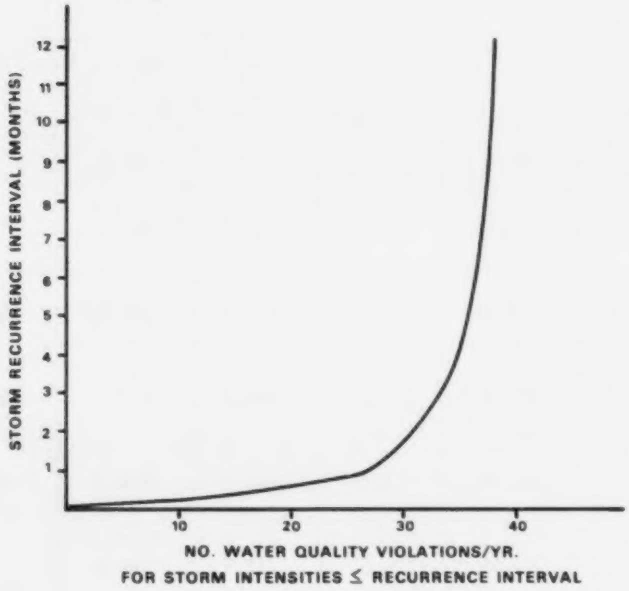


Figure 6. Water Quality (Fecal Coliform) Violations Vs. Storm Recurrence Interval - Lake Onondaga, Syracuse, New York

At Lake Washington, human viruses were readily detected in CSO's (12). Based on literature indicating that viruses present at detectable levels can cause infection CSO may be a health hazard for people using Lake Washington beaches soon after overflows.

Nutrients

Surface water bodies with long detention times, such as lakes and estuaries, tend to concentrate nutrients, organics, and metals in both supernatant and bottom muds. Under anaerobic conditions, boat mixing, concentration changes, and others, these pollutants can be resuspended and become available for plant growth or a depletion of DO.

In Lake Eola, Florida, urban runoff was found to be the sole source of lake degradation (13). Figure 7 depicts the Lake Eola watershed. Urban runoff is the only flow entering the lake. Phosphorous concentrations in the runoff were found to significantly increase algal productivity. Table 2 indicates the lake's algal growth response to urban runoff. Based on the first phase recommendations of the SCS Program's project, the 314 Clean Lakes Program is funding a phase two restoration program. It is intended to jointly fund a phase three post restoration evaluation.

Table 2. Algal Response to Urban Runoff - Lake Eola, Orlando, Florida

	Average Chlorophyll a Concentration (mg/m^3)					
	Initial	4 Days	7 Days	8 Days	9 Days	14 Days
Control	11.7	12.1	6.5	7.1	5.0	1.3
Lake Water + 5% Runoff	11.5	25.2	26.7	30.6	28.5	17.7
Lake Water + 10% Runoff	11.7	38.9	38.4	43.3	38.6	31.5
Lake Water + 25% Runoff	11.7	64.5	74.2	91.6	84.7	68.6
Lake Water + 50% Runoff	11.7	33.9	41.8	44.2	44.2	15.8
Lake Water + 75% Runoff	11.7	22.4	16.6	20.7	16.8	7.1
Lake Water + 100% Runoff	11.7	23.3	16.2	14.5	15.3	3.7

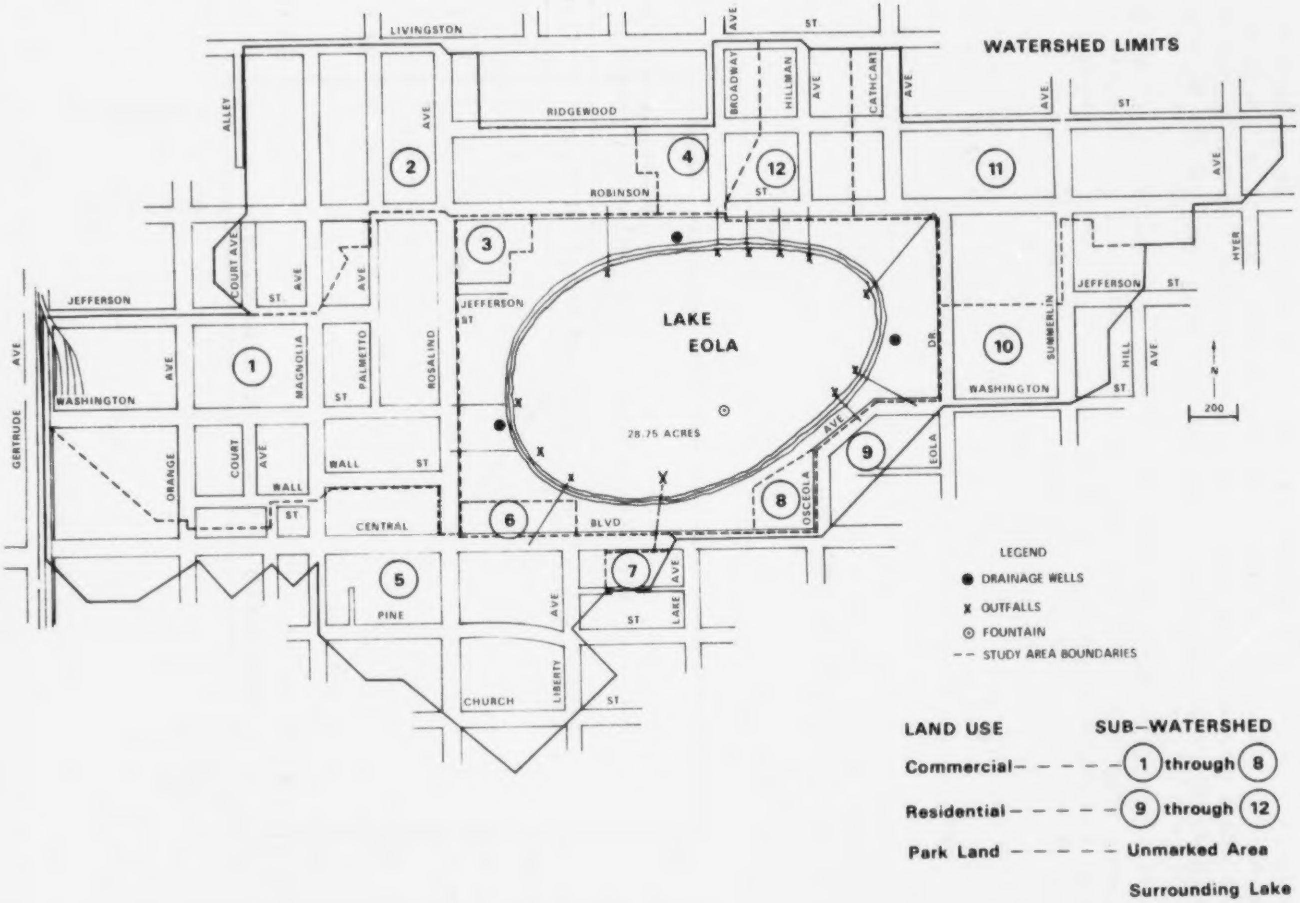


Figure 7 Lake Eola Watershed - Orlando Florida

Biological Investigations

To develop an assessment of the environmental impact of urban stormwater runoff requires a comprehensive in-depth analysis of water quality and the biological community in the receiving stream. Not only the physical and chemical characteristics of the receiving water body and its sediment should be examined, but also the aquatic and benthic organisms should be analyzed.

Such an investigation is going on in San Jose, California (14) where preliminary sampling found that the non-urbanized section of Coyote Creek supported a diverse assemblage of fish and benthic macroinvertebrates as compared to the urbanized portion which was completely dominated by pollution tolerant mosquito fish and tubificid worms. Figure 8 illustrates this point. More pollution tolerant algae also dominated the urbanized reaches of the creek. Similar results were found in the previously mentioned Lake Washington project (12) where bottom organisms (aquatic earthworms) near storm and combined sewer outfalls were more pollution tolerant relative to organisms away from these outfalls. Aquatic earthworm numbers and biomass was found to be enhanced within the zone of influence of the monitored combined sewer and storm drain outfalls in Lake Washington.

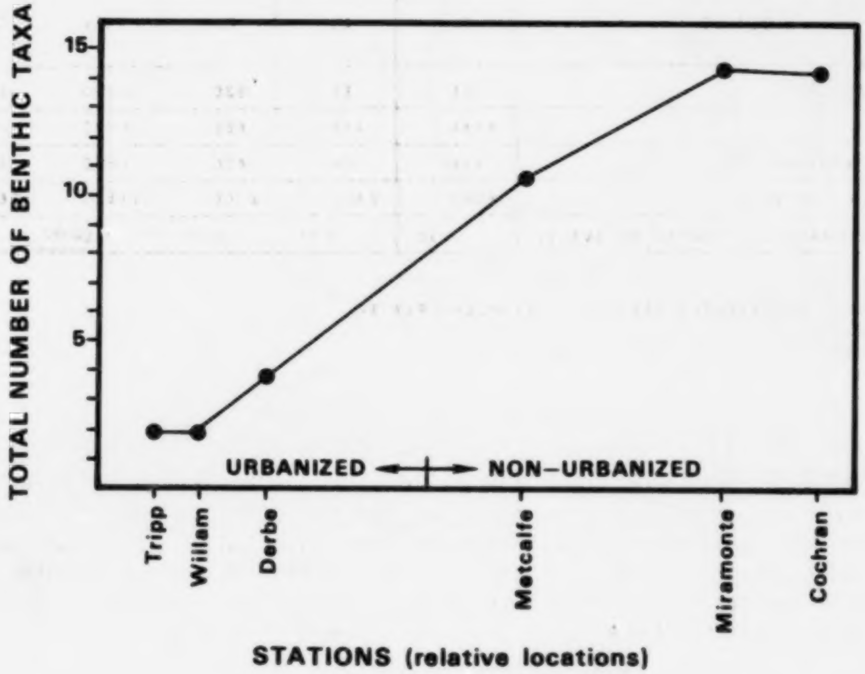


Figure 8. Abundance of Benthic Taxa - Coyote Creek, San Jose, California

For example, at one combined sewer outfall the biomass enrichment amounted to 82 percent of the background control value and aquatic earthworms constituted 90 percent of the biomass.

Toxicity

Toxicity problems can result from minute discharges of metals, pesticides and persistent organics, which may exhibit a subtle long-term effect on the environment by gradually accumulating in sensitive areas.

The SCS Program has been analyzing for toxic materials at the street surface, in urban runoff and CSO's for more than ten years. This has resulted in a large data base that identified urban runoff as a principal source of toxic pollutants. For example, New York Harbor receives metals from treatment plant effluents; combined sewer discharges; separate storm sewer discharges; and untreated wastewater (15). As seen in Table 3, urban runoff is the major contributor of heavy metals to the harbor.

Table 3. Metals Discharged to the Harbor From New York City Sources

SOURCE	Cu	Cr	Ni	Zn	Cd
PLANT EFFLUENTS	1,410	780	930	2,520	95
RUNOFF*	1,990	690	650	6,920	110
UNTREATED WASTEWATER	980	570	430	1,500	60
TOTAL WEIGHT (LB/DAY)	4,380	2,040	2,010	10,940	265
WEIGHTED AVERAGE CONCENTRATION (MG/L)	0.25	0.12	0.11	0.62	0.015

*IN REALITY, SHOCKLOAD DISCHARGES ARE MUCH GREATER.

Table 4 shows the total annual mass of selected constituents from a combined sewer and storm overflow point in Seattle (12). A high percentage of the heavy metals and toxic materials is associated with the suspended solids or particulates which may tend to concentrate in the sediments. This can help us in terms of treatment since it is easier to separate pollutants attached to suspended solids.

Table 4. Total Vs. Particulate Mass From a Combined Sewer and Storm Sewer Overflow Point - Lake Washington, Seattle, Washington

	SELECTED CSO POINT		SELECTED STORM DRAIN POINT	
	TOTAL* MASS(KG)	PARTICULATE* MASS(KG)	TOTAL* MASS(KG)	PARTICULATE* MASS (KG)
SUSPENDED SOLIDS	3920	3920	2238	2238
Cu	1.33	1.04	1.16	.745
Hg	.026	.023	ND	ND
Pb	1.46	1.01	6.04	5.32
Zn	3.20	2.20	2.74	1.76
Al	66.0	63.4	97.2	93.9
ORGANIC C	458	193	299	168
TOTAL P	45.1	12.8	8.73	4.60
OILS AND GREASES	330	NA	113	NA
CHLORINATED HC	ND	.460G	ND	.388G

NA: NOT APPLICABLE ND: NOT DETERMINED

*KG=2.2 LB.

Sediment samples were analyzed for metals, organic carbon, phosphorous, chlorinated hydrocarbons and polychlorinated biphenols (PCB's). As can be seen from Figure 9, a composite index to assess wet-weather impacts was 16 times the minimum background control value. Also, pesticide levels in sediments along the Seattle shoreline of Lake Washington were up to 37 times background concentrations. The measured pesticide content of the sediments was predominantly DDT and its degradation products. Although DDT is no longer used this flags a potential problem with the use of newer toxic chemicals and of their long-term fate and transport.

In the previously mentioned San Jose project (14) urban sediment compared to non-urban sediment from Coyote Creek contained higher concentrations of lead (10 times greater), arsenic (9 times greater) BOD_5 (up to 4.4 times greater) and ortho phosphates (up to about 4 times greater). Significantly greater concentrations of high molecular weight hydrocarbons and oxygenated compounds were also found in the urban samples.

Lead concentrations in urban samples of algae, crawfish and cattails were two to three times greater than in non-urban samples, while zinc concentrations were about three times the non-urban concentrations. Bioaccumulation of lead and zinc in the organisms compared to water column concentrations was at least 100 to 500 times greater.

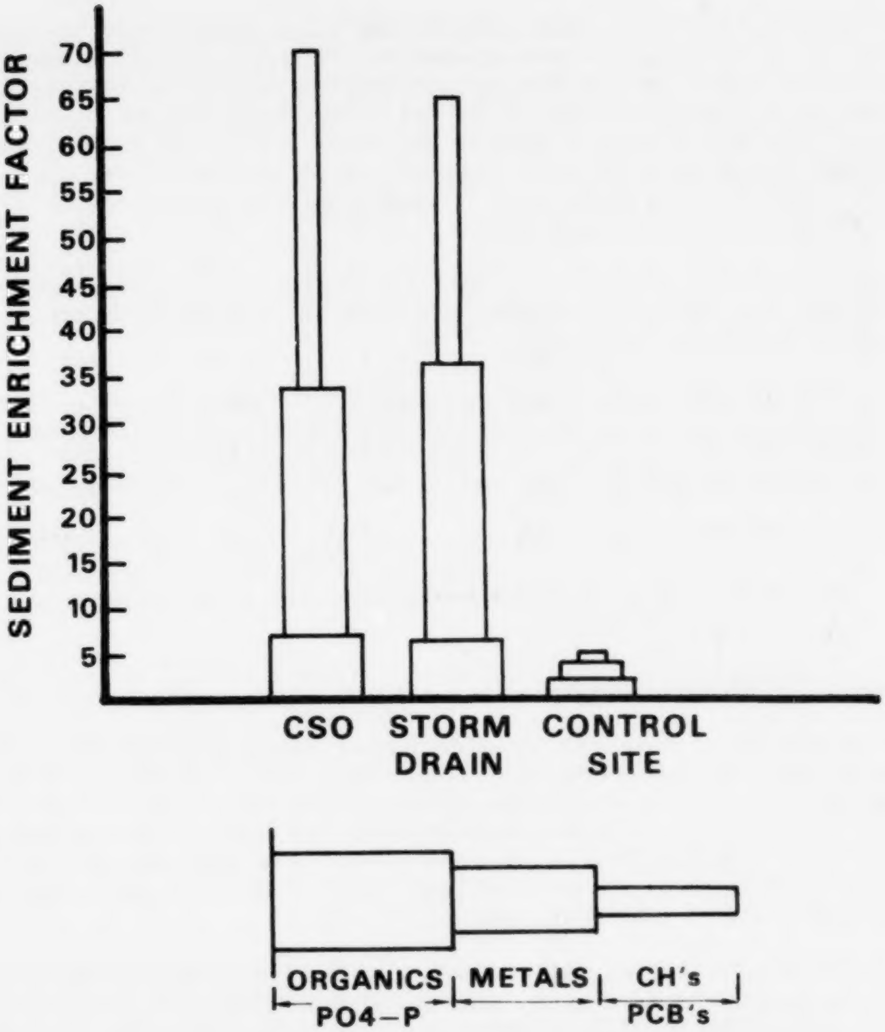


Figure 9. Illustration of Urban Sediment Enrichment - Lake Washington, Seattle, Washington

Petroleum enters urban runoff from many dispersed sources -- motor vehicles being the most significant. It may be attributed to spills, leaks and disposal of motor vehicle lubricants, antifreeze and hydraulic fluids.

Petroleum hydrocarbons, particularly the polynuclear aromatics have been shown to be carcinogenic and mutagenic. At New York City's Newtown Creek treatment plant (15) 24,000 gallons (90,840 liters) of oil and grease, [equivalent to a moderate harbor spill (16)] were bypassed during one four hour storm. The SCS Program's Jamaica Bay study (17) found that 50 percent of the hexane extractable material contributed to the Bay is due to CSO's. It is clear that further investigation into receiving water impacts due to urban runoff warrants consideration.

A recently funded project (18) with Onondaga County is screening urban runoff and CSO for bacterial mutagens and is assessing the mutagenic/carcinogenic potential of the toxicants.

Recently, the SCS Program arranged with EPA's Region II Surveillance and Analysis Division to analyze for priority pollutants in urban samples supplied from ongoing projects. Results from a few locations show that a significant amount of priority pollutants are present in urban runoff.

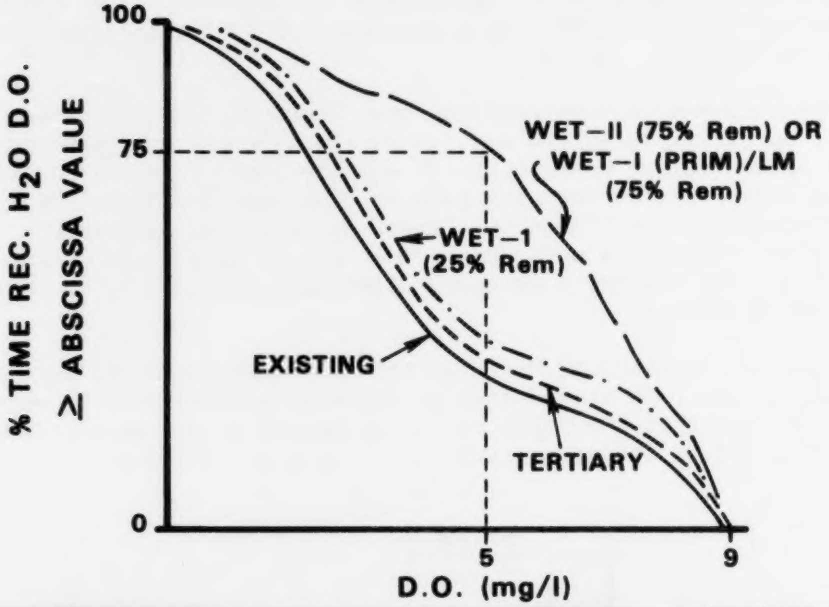
In FY80, the Program is looking to supplement this with extramural assistance. In any case, in all appropriate projects, priority pollutant runs are taken to initially identify toxic substances for source identification and source control studies.

CONTROL METHODOLOGY BASED ON IMPACTS

The optimal goal of an urban stormwater pollution control study is to evaluate the relative impact of runoff and CSO's on the receiving waters and decide what control alternatives would be most cost effective in reducing wet-weather pollution. Impacts are often site specific and the extent of the problems will depend heavily on local conditions, such as rainfall quantities, point sources of pollution and their treatment, land use, and the sensitivity of the receiving water.

In an effort to catalog existing data, case studies of documented receiving water impacts are being reviewed in a project with the University of Florida (19). The project is expected to identify the types of impacts requiring further verification and the research needed to quantify receiving water stresses.

The concept for a simplified continuous receiving water quality model has been developed during the nationwide evaluation of CSO's and urban stormwater discharges project (7) and refined into a user's manual by a subsequent project (20). This model, termed Level III - Receiving, permits preliminary planning and screening of areawide wastewater treatment alternatives in terms of frequency of water quality violations and using more traditional approaches such as DO profiles. Figure 10 represents the type of analysis facilitated by the model.



CONTROL ALTERNATIVES	% BOD REMOVAL		COST ($\$\times10^6$)
	DRY WEATHER	WET WEATHER	
EXISTING	85	0	–
TERTIARY	95	0	6
WET–I (PRIMARY)	85	25	1
WET–II (ADV)	85	75	6
WET–I/LAND MGMT.	85	75	3

Figure 10. Hypothetical Example Solution Methodology

By this analysis a truer cost-effectiveness comparison can be made based on total time of impact to receiving water and associated abatement costs. For example, if a 5 mg/l DO is desired in the receiving water 75 percent of the time, an advanced form of wet-weather treatment or primary wet-weather treatment integrated with land management is required. The latter is the most cost-effective at $3M. This can help set cost-effective standards as well as select alternatives.

Also a general methodology has been developed for evaluating the impact of CSO's on receiving waters and for determining the abatement costs for various water quality goals (11). It was developed from actual 201 facility planning experience in Onondaga Lake in Syracuse, New York. An important goal of studies to determine the impact of waste discharges on a receiving water is to predict the waste loads that can be assimilated without violation of water quality standards so that a loading curve such as shown in this figure can be defined.

Figure 11 illustrates the potential effect storm loads may have in violating a 5 mg/l DO standard after dry-weather treatment is upgraded. It further implies that CSO pollution loads should be abated next, since they are the easiest of the storm loads to control and capture.

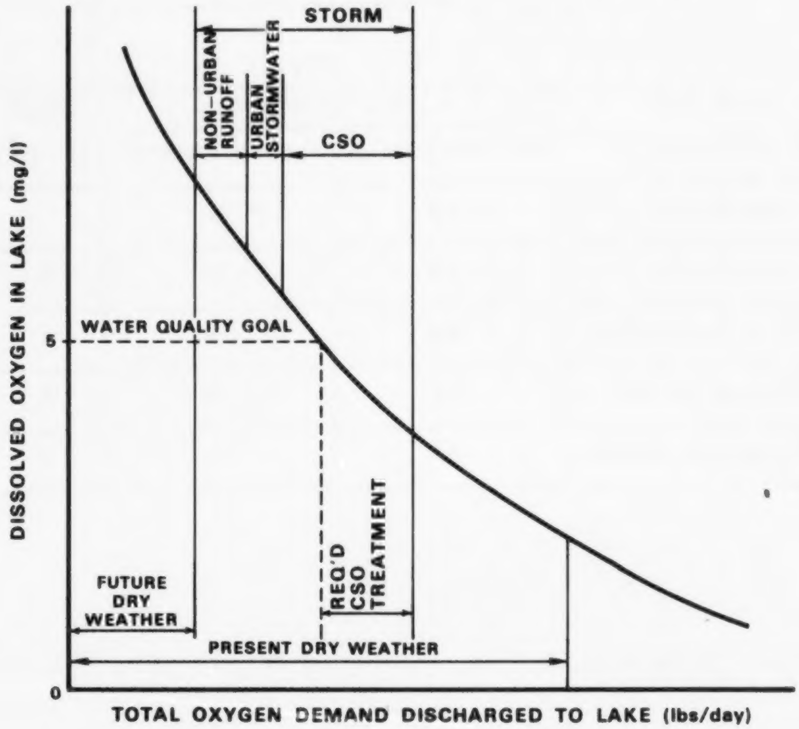


Figure 11. A Typical Loading Curve Relating Pollutant Load to Water Quality Response

The methodology outlines the interrelationships of various phases of a CSO study and develops procedures necessary to arrive at the costs to meet various water quality goals.

A project has been initated with Manhattan College (21) to develop a methodology for defining criteria for wet-weather water quality standards. The methodology will provide assistance in determining the degree of treatment of intermittent urban discharges. Recognizing an important gap in the aforementioned methodologies the duration of water quality standards violations vs. species survival will be taken into consideration. This may be difficult to accomplish since there is a severe lack of data in this area.

The research will include the definition and testing of a method for evaluating wet-weather water quality criteria considering both classical pollutants and toxics for the different types of receiving waters. Load matrices will be developed for determining the load of a constituent as a function of urban area size and the size and frequency of wet-weather events. These arrays will act as input to effects matrices.

Other factors influencing the effects matrices are the receiving water size and type and the organism/contaminant transformations. Thus, situations where water quality problems associated with discharges from urban runoff will be defined. An effort will be made to generalize the results on anticipated problem solutions to yield a national scale assessment of the magnitude of the problem.

This project will coordinate with other ongoing receiving water impact studies both within and outside the SCS Program.

A user's manual/guidebook for wet-weather flow quality and quantity monitoring is currently being developed (22). There are various reasons for entering into a wet-weather sampling and monitoring program and based on the desired objectives the approach philosophy will vary. This manual will delineate various approach philosophies to statistically satisfy various wet-weather sampling/monitoring needs/objectives, including separate sections on best management practice (BMP) evaluation methodologies and receiving water impact methodologies to assure statistical reliability and transferability of data within these areas.

CONCLUSION AND RECOMMENDATIONS

Under certain conditions storm runoff can govern the quality of receiving waters regardless of the level of dry-weather flow treatment provided. Hence, control of runoff pollution can be a viable alternative for maintaining receiving water quality standards.

The SCS program has had only partial success in finding urban storm flow generated receiving water impacts employing the conventional DO parameter. The problem appears to be in the use of conventional dry-weather monitoring applied for unsteady-state flow regimes caused by storms. Based on a comparative analyses of wet vs. dry-weather oxygen demanding substances loads there remains a high potential for these impacts to occur in receiving waters. Further studies should be considered to resolve the anomaly between actual runoff loadings and observed receiving water impacts. The Program has been more successful in sediment analysis than in water column analysis for finding DO depletions. Direct evidence has been obtained [from the Milwaukee River project (9)] of how a disturbed benthos depletes DO from overlying waters. Studies have also shown that storm and CSO adversely affect sediment by toxics enrichment and resultant biological upsets. Since particulate matter in untreated storm and CSO is larger, heavier, and in significant quantities when compared to treated sanitary effluent more needs to be known about the fate and transport of settleable and separable materials. Hydrodynamic solids separation and sediment transport routines must be added to receiving water models to take care of the neglected or presently omitted but significant particulate and bed load flow fields.

Urban runoff has been identified as a major source of toxic substances including heavy metals, which have been shown to concentrate in biological species and sediment, and petroleum hydrocarbons some of which are known carcinogens and matagens. Very little is known about the long term effects of these substances. Further investigation into their receiving water impacts is needed. Integration of various research program efforts and results are needed in this area to optimize the benefits, avoid duplications and minimize the costs involved.

In the area of control methodology, a simplified continuous receiving water quality model has been developed for preliminary planning and screening of areawide wastewater treatment alternatives in terms of frequency of water quality violations. Also a general methodology has been developed for evaluating the impact of CSO's on receiving waters and for determining the abatement costs for achieving various water quality goals. Still more work is needed on methodologies.

Mandates of the law are upon us, including wet-weather flow pollution control; monies are being spent at large scale by EPA and others for water pollution cleanup. In order for governments to execute their function in this area properly, wet-weather flow pollution must be considered, and research and development be fostered to back this need.

REFERENCES

1. Olivieri, V.P., et al. Microorganisms in Urban Stormwater. EPA-600/2-77-087, U.S. Environmental Protection Agency, 1977, NTIS No. PB 272 245.

2. Lager, J.A., et al. Urban Stormwater Management and Technology: Update and Users' Guide. EPA-600/8-77-014, U.S. Environmental Protection Agency, 1977, NTIS No. PB 275 654.

3. Keefer, T.N., et al. Dissolved Oxygen Impact from Urban Runoff. EPA-600/2-79-156, U.S. Environmental Protection Agency, 1979, NTIS No. Pending.

4. Stiefel, R.C. Dissolved Oxygen Measurements in Ohio Streams Following Urban Runoff. EPA Draft Final Report, EPA Grant No. R-805201.

5. Ketchum, L.H., Jr. Dissolved Oxygen Measurement in Indiana Streams During Urban Runoff. EPA-600/2-78-135, U.S. Environmental Protection Agency, 1978, NTIS No. PB 284 871.

6. McConnell, J.B. Impact of Urban Storm Runoff on Stream Quality Near Atlanta, Georgia. EPA Draft Final Report, EPA Inter-Agency Agreement No. IAG-D6-0137.

7. Heaney, J.F., et al. Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges, Volume II - Cost Assessment and Impacts. EPA-600/2-77-064b, U.S. Environmental Protection Agency, 1977, NTIS No. PB 266 005.

8. Field, R. and D.C. Ammon. Potential of Stormwater Impacts Based on Comparative Analysis of Wet and Dry Weather Pollutant Loading. In: paper presented at National Conference on Urban Stormwater and Combined Sewer Overflow Impact on Receiving Water Bodies, Orlando, Florida, November 26-28, 1979.

9. Meinholz, T.L., et al. Verification of the Water Quality Impacts of Combined Sewer Overflows. EPA-600/2-79-155, U.S. Environmental Protection Agency, 1979, NTIS No. Pending.

10. Klemetson, S.K., et al. Movement and Effects of Combined Sewer Overflow Sediments in Receiving Waters. EPA Draft Final Report, EPA Grant No. R-806111.

11. Moffa, P.E., et al. Methodology for Evaluating the Impact and Abatement of Combined Sewer Overflows: A Case Study of Onondaga Lake, New York. EPA Draft Final Report, EPA Grant No. R-805096.

12. Tomlinson, R.D., et al. Fate and Effects of Sediments from Combined Sewer and Storm Drain Overflows in Seattle's Nearshore Waters. EPA Draft Final Report. EPA Grant No. R-805602.

13. EPA Grant No. R-805580. Stormwater Management to Improve Urban Lake Water Quality. University of Central Florida, Orlando, Florida.

14. Pitt, R. and M. Bozeman. Water Quality and Biological Effects on Urban Runoff on Coyote Creek. EPA Draft Final Report, EPA Grant No. R-805418.

15. Mytelka, A.I., et al. Combined Sewer Overflow Study for the Hudson River Conference. EPA-R2-73-153, U.S. Environmental Protection Agency, 1973, NTIS No. PB 227 341.

16. Congressional Federal Register. National Oil and Hazardous Substances Pollution Contingency Plan. Title 40, Chapter 5, Part 1510, Subpart A, Section 1510.5(1)(2) (Definition of Medium Discharge: 10,000 to 100,000 gal. oil to costal waters), July 1, 1976.

17. Feuerstein, D.L. and W.O. Maddaus. Wastewater Management Program, Jamaica Bay - Volume I: Summary Report. EPA-600/2-76-222a, U.S. Environmental Protection Agency, 1976, NTIS No. PB 260 887.

18. EPA Grant No. R-806640. Evaluation of Urban Runoff and Combined Sewer Overflow Mutagenicity. Onondaga County Department of Drainage and Sanitation, Syracuse, New York.

19. EPA Grant No. R-805663. Nationwide Assessment of Receiving Water Impacts from Urban Stormwater Pollution. University of Florida, Gainesville, Florida.

20. Medina, M. Level III: Receiving Water Quality Modeling for Urban Stormwater Management. EPA-600/2-79-100, U.S. Environmental Protection Agency, 1979, NTIS No. Pending.

21. EPA Grant No. R-806828. Development of Methods to Define Water Quality Effects of Urban Runoff. Manhattan College, Bronx, New York.

22. EPA Grant No. R-806345. Prepare Guidelines for Wet-Weather Monitoring. Urban Drainage and Flood Control District, Denver, Colorado.

3

Second Session

IMPACTS ON LAKES

Moderator: Waldron M. McLellon
University of Central Florida
Orlando, Florida

AN APPROACH FOR ASSESSING THE WATER QUALITY SIGNIFICANCE OF CHEMICAL CONTAMINANTS IN URBAN LAKES

G. Fred Lee and R. Anne Jones
Colorado State University
Fort Collins, Colorado

ABSTRACT

The conversion of rural lands to urban area is often accompanied by a significant increase in the total amount of phosphorus and nitrogen derived per unit area of a waterbody's watershed. For rural lands phosphorus without livestock, export coefficients typically range from 0.005 to 0.5 g $P/m^2/yr$, while in urban areas the typical export coefficient is 0.1 g $P/m^2/yr$. A significant part of this difference is due to the much greater water yield per unit area of watershed in urban areas compared to rural areas.

Urban runoff typically contains appreciable quantities of both soluble ortho P and particulate forms of P. Studies have been conducted to evaluate the amounts of available forms of phasphorus present in typical urban stormwater drainage for several municipalities located across the U.S. It has been found that on the order of 10 to 30 percent of the particulate phosphorus present in urban stormwater drainage would likely become available to affect algal growth in a lake or stream. As a result of these findings, the focal point of the control of nitrogen and phosphorus from urban stormwater sources should be directed toward the soluble orthophosphate component. Most stormwater drainage control programs are directed toward control of particulate matter. Such programs are likely to have limited impact on eutrophication-related water quality in urban lakes, since only a small part of the particulate phosphorus will likely become available to stimulate aquatic plant growth in the waterbody.

The OECD (Organization for Economic Cooperation and Development) eutrophication modeling study which included about 40 waterbodies across the U.S. and 200 waterbodies in Western Europe, North America, Japan, Australia, etc., has shown that the phosphorus load normalized by waterbody mean depth and hydraulic retention time is correlated to the planktonic algal chlorophyll concentration, planktonic algal-related water clarity and hypolimnetic oxygen depletion rate. The results of the OECD study provide the tools necessary to quantitatively assess what water quality improvement can be achieved as the result of various nutrient control efforts.

INTRODUCTION

Increasing attention is being given to management of water quality in urban lakes. While these waterbodies have previously been generally neglected by pollution control agencies, the US EPA and state and local governmental agencies are beginning to provide funding to investigate sources of contaminants for urban lakes and to develop water quality control programs for them. Two of the major forces behind these efforts are the US EPA Urban Lakes and the Combined Sewer Overflow Programs. These programs, if carried out properly, will provide municipalities with the information needed to develop cost-effective, technically valid water quality management programs for urban lakes. This paper considers a number of factors that must be evaluated in the development of such programs.

CHARACTERISTICS OF URBAN LAKES

The first issue that must be addressed in the development of urban lakes' water quality management strategy is whether or not urban lakes behave differently from rural lakes with respect to contaminant load-water quality response. Generally urban lakes tend to be shallow, with maximum depths on the order of a couple of meters. With few exceptions they are highly fertile and would generally be classified as eutrophic. The littoral areas of many urban lakes are dominated by macrophytes and attached algae. They normally have "pea soup" green to brown color arising from planktonic algae and suspended sediment. Occasionally they will have dense algal blooms which can create obnoxious surface scum. Thermal stratification, if it develops, is usually short-lived; rarely would a thermocline persist throughout the summer growing season. With few exceptions, urban lakes have appreciable populations of rough fish such as carp and bullheads. All of these characteristics tend to make urban lakes fall into the group of shallow, weedy, moderately-to-highly turbid waterbodies which have appreciable mixing between the sediments and overlying waters. The overall water quality in urban lakes would normally be classified as poor; it is generally believed that if the public had a choice, they would recreate in rural waterbodies. However, many urban lakes have greater fishing and other recreational pressures placed on them than their rural counterparts. This pressure will likely increase significantly as the energy and economic situation continues to deteriorate. It is therefore imperative that an understanding of the factors controlling urban lake water quality be achieved in order to maximize water quality improvements for funds spent in control programs.

In those municipalities where salt is used for road deicing, the first major snow melt in the spring can bring appreciable amounts of salt into the lake. Some urban lakes have become meromictic (temporarily or permanently stratified) because of this problem.

In addition to water quality problems there is also the aesthetic problem of littering of the shores and bottom of urban lakes by the public. Unless effectively policed, the littering can be a significant deterrent to the use of urban lakes.

While most urban lakes are freshwater, in coastal areas some are marine-estuarine which may be subject to the influence of the tide. In some municipalities urban lakes are dammed rivers and therefore may have appreciable contaminant input from upstream sources. Many urban lakes are used as receptacles for combined sewer overflow in municipalities where combined sewers exist. Routinely they are the receptacle for urban stormwater drainage and frequently they receive domestic wastewater inputs from separated sewerage systems during periods of high flow, electrical outages, pump failure, etc.

WATER QUALITY MANAGEMENT IN URBAN LAKES

WATER QUALITY PROBLEM IDENTIFICATION

The first step in the development of a water quality management program for any waterbody is an assessment of the water quality problems that exist within the waterbody. Generally, the most significant and in many cases the only water quality problem that exists in urban lakes is excessive fertilization. While many urban lakes may receive relatively large loadings of a variety of toxic chemicals such as heavy metals, pesticides, etc., it appears that these contaminants rarely cause significant water quality problems in urban lakes. This is a result of the fact that more eutrophic waterbodies are able to "detoxify" many contaminants as a result of contaminant interaction with suspended and deposited particulate matter.

Urban lakes tend to receive relatively large loads of oxygen demanding materials in urban stormwater drainage and combined sewer overflows and/or domestic wastewater bypasses to the lakes. While oxygen demand problems can occur in urban lakes, they tend to be infrequent. The overall trophic status of many of these waterbodies is such that rarely would low dissolved oxygen be a significant cause of water quality deterioration during ice-free periods. However, as discussed by Lee and Jones (1979c), in the winter in areas where waterbodies develop an ice cover, shallow lakes of this type, especially during periods of snow cover preventing penetration of light through the ice, tend to be prone to winterkill where more desirable game fish would be lost because of low dissolved oxygen.

Urban stormwater drainage and the entrance of domestic wastewaters into urban lakes tends to cause elevated coliform counts which can serve as a basis for closing beaches and restricting other body contact with the water. Urban lakes in regions where there is appreciable new construction can receive large amounts of erosional materials which will result in more rapid siltation especially near the point where the stormwaters enter the waterbody. These situations can cause water discoloration due to the presence of large amounts of suspended sediment in the urban stormwater drainage.

While it is possible to generalize on water quality problems of urban lakes, the water quality problems of any particular urban lake must be assessed on a case-by-case basis in which deterioration of water quality is judged based on impairment of a beneficial use of the water. Water quality should never be judged based on the concentration of a chemical contaminant in the water or the sediment. Many pollution control agencies make a significant error in assuming that one can relate water quality to the chemical composition of the water as determined by standard chemical analytical procedures normally used in the water quality field. Chemical contaminants exist in aquatic systems in a variety of chemical and physical forms, only some of which are available to affect water quality. Basing an urban lake water quality control program on the total contaminant load or concentration for a particular contaminant can easily result in the public spending large amounts of money in the name of water pollution control with little or no improvement in urban lake water quality. As indicated above, those responsible for developing water quality management programs for urban lakes must first assess and document how the public's use of the urban lake is impaired by a particular water quality characteristic. Once this is known, it is then possible to begin to formulate the studies that are needed to provide the information which can serve as a basis for developing a water quality management strategy for a particular waterbody.

WATER QUALITY CRITERIA AND STANDARDS

The traditional approach that has been followed in the water quality control field during the past 15 years and has been used as a basis for developing water quality management programs has involved the use of water quality criteria and/or standards. Generally fixed numeric values have been established against which results of chemical analyses of water and/or sediments have been compared. This approach has worked reasonably well for the control of water quality deterioration arising from many municipal and industrial point source contaminants, where the objective of the control program was the elimination of gross water quality deterioration such as aquatic organism acute toxicity. Public Law 92-500, the 1972 Amendments to the Water Pollution Control Act, requires that the US EPA develop water quality criteria which are to become the backbone of the water pollution control program in the U.S. The US EPA released the first set of these criteria, commonly called the "Red Book" criteria, in July, 1976 (US EPA, 1976). Another set is currently under review. Public Law 92-500 requires that every state review its existing water quality standards every three years and that standards acceptable to the US EPA be promulgated by each state. Some states are adopting, without modification, US EPA July, 1976 criteria as fixed numeric standards. These criteria are "worst case" values developed by exposing organisms for chronic - usually life-time periods of time to forms of the contaminants which are 100% available. They are designed to protect fish and other aquatic life against some of the more subtle effects of contaminants such as impairment of reproduction and growth rate. For many chemicals there is a several order of magnitude difference between the critical concentration which will kill fish within 96 hours and the chronic safe concentration. Adoption of the US EPA July, 1976 Red Book criteria as water quality standards will, in general, provide the basis for ecologically protective water quality management programs. It is likely that there will be few instances where a waterbody containing concentrations of contaminants equal to or less than the Red Book criteria numeric concentrations would experience deteriorated water quality due to chemicals considered in the Red Book.

While ecologically protective, there are many who question the cost-effectiveness of application of standards numerically equal to the worst case criteria since that approach does not consider the fact that substantial parts of many chemical contaminants present in aquatic systems are in chemical forms which are unavailable to affect aquatic organisms and water quality. This is especially true for contaminants associated with particulate matter. Lee et al. (1978a), Jones and Lee (1978b), and Jones et al. (1979) have conducted a comprehensive study of the water quality significance of chemical contaminants associated with aquatic sediments from locations across the U.S. It has been found that most of the contaminants associated with aquatic sediments are in forms unavailable to affect water quality. They have pointed out that there is no relationship between the total concentration of a contaminant in a sediment and the potential for this contaminant to affect water quality. They further pointed out that it is not appropriate to use water quality standards numerically equal to the Red Book criteria for many aquatic systems, especially those which contain large amounts of sediment-associated contaminants. Since the behavior of contaminants in urban lakes is frequently greatly influenced by sediments both suspended and deposited, application of water quality standards equal to Red Book criteria would not in most instances be a cost-effective approach for developing a water quality management strategy for these systems. Since the primary sources of contaminants for urban lakes are urban stormwater drainage and in some instances combined sewer overflow and domestic wastewater diversions, and since numerous studies have found that appreciable parts of the contaminants in these sources are in particulate forms, it is likely that water quality programs directed toward the total contaminant load to an urban lake could readily result in expenditure of large amounts of money for contaminant control with little or no improvement in urban lake water quality. Because of the relatively high cost associated with the control of contaminants in urban stormwater drainage, it is imperative that all funds spent for this purpose be directed to the maximum extent possible to the control of available forms of contaminants.

HAZARD ASSESSMENT APPROACH

General Aspects*

In situations where chemical water quality standards numerically equivalent to the US EPA Red Book criteria are not suitable indicators of water quality deterioration, such as in assessing the water quality impact of contaminants in urban stormwater drainage on urban lakes, it is necessary to use the hazard assessment approach to develop a technically valid, cost-effective, ecologically protective program for water quality management. This approach was originally developed for prescreening new chemicals for their environmental impact. Several papers in the book edited by Cairns *et al*. (1978) described the application of this approach for this purpose. Lee and Jones (1979a,b) and Lee *et al*. (1979b) have described the application of this approach for existing chemicals. Pollution control agencies will have to follow an approach similar to that discussed by Lee and Jones (1979a,b) and Lee *et al*. (1979b) in developing water quality control programs for urban lakes if these programs are to maximize water quality improvement for the money spent.

An aquatic environmental hazard assessment is built on two basic components: aquatic toxicology and environmental chemistry-fate information. In the aquatic toxicology phase, the concentrations of the contaminant, effluent, or other source (such as urban stormwater drainage) of concern which cause no adverse impact on aquatic organisms is established. As part of this evaluation, concentration-duration of exposure couplings are developed for each contaminant or potentially hazardous input which define the "no adverse effect" level for a spectrum of exposure durations which may be encountered in the aquatic environment. Figure 1 shows a general-case concentration-duration of exposure relationship. It shows that for short durations of exposure, organisms can tolerate higher concentrations of contaminants without being adversely affected. As the exposure duration increases, the no effect concentration decreases eventually to a chronic safe level. For particular contaminants, these couplings must be based on concentrations of available forms; for a specific effluent containing a mixture of potentially hazardous materials, this can be based on dilution of effluent or other source of contaminants. In the aquatic toxicology phase of hazard assessment the bioaccumulation potential of the contaminant is also assessed. Assessment of this potential impact will be discussed further in a subsequent section.

---

*This discussion is adapted from Lee and Jones (1979b).

The environmental chemistry-fate portion of an environmental hazard assessment considers the chemical processes that occur within the aqueous system as well as interactions with other environmental components that alter the form of the contaminant of concern and its transformation products in the aquatic system. Major types of chemical reactions that commonly occur in the aquatic environment which could cause significant changes in the form of a potentially hazardous chemical in terms of its water quality impact are acid-base, precipitation, complexation, oxidation-reduction, hydrolysis, photolysis, gas transfer, biochemically mediated reactions - biotransformation, and sorption, both biotic and abiotic. The environmental chemistry-fate phase of a hazard assessment for an aquatic system also determines the transport pathways of the contaminant and its transformation products from the point at which it enters the aquatic system to its final disposition or the point at which it leaves the system. In association with this, the physical processes of advection-transport, and dilution-dispersion must be defined for the terrestrial, atmospheric, and aquatic environments with which the contaminant of concern and its transformation products come in contact. All of the pertinent environmental chemistry and fate information is formulated into a series of differential equations which describe each potentially significant transformation and transport pathway. Such an environmental chemistry-fate model is described schematically in Figure 2. Once an environmental chemistry-fate model has been verified it can be used to predict for a particular contaminant input, the concentration of the contaminant of concern and each potentially significant transformation product in each aquatic environmental component, i.e., in solution, associated with particulates, and associated with organisms, etc.

During the process of making an assessment of the environmental hazard associated with the particular source of contami-

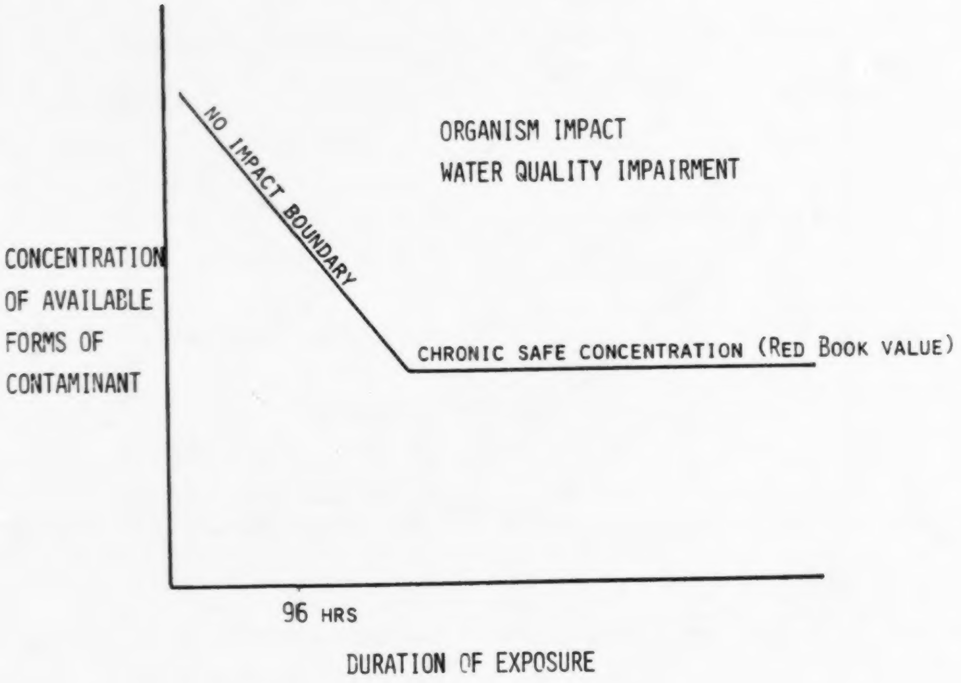
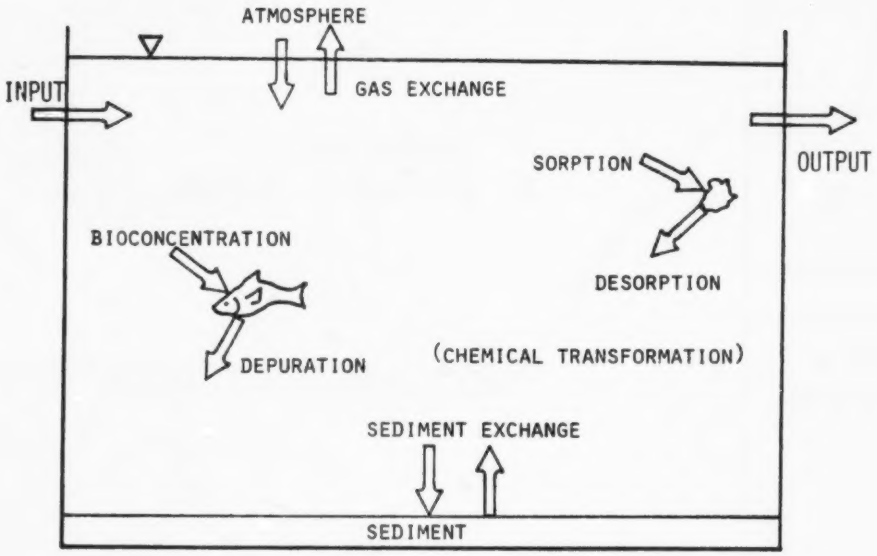

NO IMPACT BOUNDARY
ORGANISM IMPACT
WATER QUALITY IMPAIRMENT
CONCENTRATION OF AVAILABLE FORMS OF CONTAMINANT
CHRONIC SAFE CONCENTRATION (RED BOOK VALUE)
96 HRS
DURATION OF EXPOSURE

FIGURE 1 - General Available Concentration - Duration of Exposure Relationship.



$$\frac{D(\text{AVAIL. FORM})}{DT} = K_1(\text{GAS EXCHANGE}) + K_2(\text{BIOCONCENTRATION}) + K_3(\text{SORPTION}) + K_4(\text{CHEMICAL TRANSFORMATIONS}) + K_5(\text{ETC.})\ldots$$

FIGURE 2 - Environmental Chemistry - Fate Model

nants, the predicted concentration-duration of exposure coupling is compared with the "no effect concentration-duration of exposure" relationship developed for the chemical or source in question in the toxicological portion of the hazard assessment (Figure 1). Figure 3 shows schematically a variety of the possible couplings and their relationship to the area of "impact." The hatched area in this figure represents an area in which there is sufficient duration of exposure to sufficient concentrations of available forms of contaminant to have an adverse effect on aquatic organisms and/or water quality. This relationship must be defined for each contaminant or source of contaminants of concern in formulating a hazard assessment. The numbered curves (1-5) in Figure 3 show results that could be obtained through environmental chemistry-fate modeling, where a combination of dilution and chemical reactivity bring about a certain concentration-duration of exposure relationship. Curve 1 represents that coupling typical of spill situation, where there is toxicity for a short time associated with the point of entry before any reactions or dilution takes place. This might also be the situation associated with the mixing zone for a particular discharge such as urban stormwater drainage or combined sewer overflow.

Curve 2 in Figure 3 is a case where the contaminant at levels that are normally found in the environment does not show any acute toxicity but is chronically toxic either to an organism or to higher forms that may use the organism as food. PCB's, DDT, and mercury would all fall into this category. However, because of concentration of some of these types of chemicals within the higher trophic level fish, there is a potential for harm to man and other animals that use these fish as a source of food.

Curve 3 in Figure 3 represents the type of situation which might be associated with municipal wastewater discharges which contain ammonia, where for short durations of exposure there is no impact because possible durations of exposure are too short. However, there is an intermediate zone at some distance from the point of discharge where expected durations of exposure of the organisms to available forms are sufficient, where there could be toxicity to fish or other organisms that reside in the area. Eventually the ammonia would be oxidized or diluted to non-toxic levels as one proceeds further down the stream from this zone.

Curve 4 is representative of the situation where there is a transformation of the contaminant added to the system which causes it to be more toxic as it goes downstream. Eventually it is either diluted or detoxified through other reactions. An example of this type of situation is one involving the addition of a complexed heavy metal to the environment where the complex is biodegradable, releasing the heavy metal at some distance downstream in sufficient concentrations to be toxic to aquatic

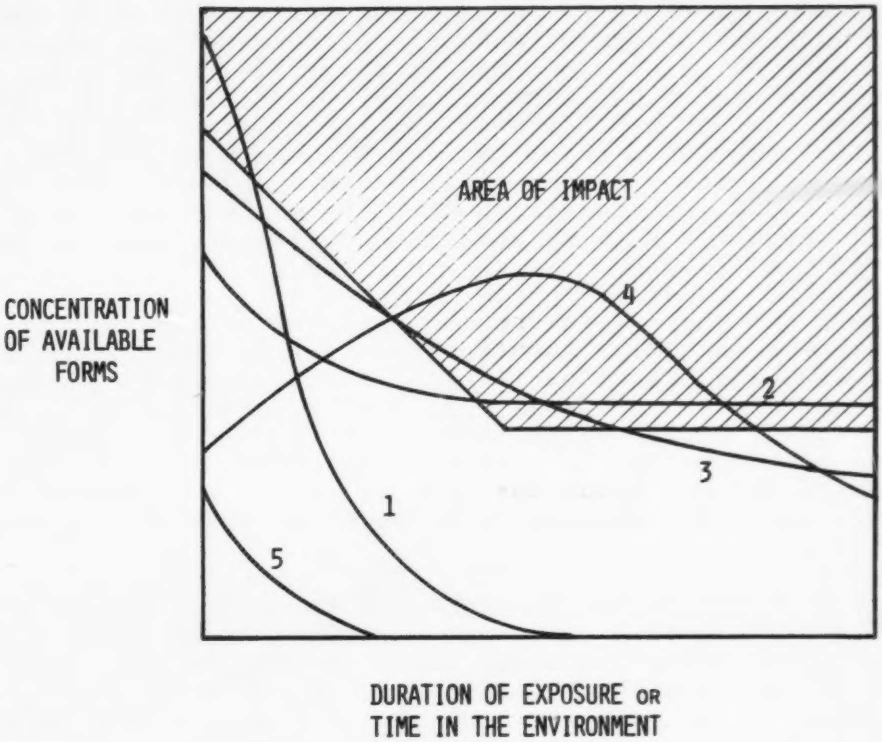


FIGURE 3 - Examples of Concentration of Available Forms - Duration of Exposure Couplings Compared to Area of Water Quality Impact

life in that region. Curve 5 is the case that exists for most chemicals for which there is sufficient treatment or controlled use so that there is no toxicity associated with it, either acute or chronic.

An environmental hazard assessment for any contaminant source should be conducted in a series of levels or tiers in which the aquatic toxicology and environmental chemistry-fate are determined with increasing sophistication and reliability with succeeding tiers, with a decision point at the end of each tier. The decision choices are: 1) do not allow discharge because of excessive expected hazard, 2) restrict discharge through use or treatment to reduce environmental hazard to an acceptable level, 3) proceed with discharge as is - expected environmental hazard is acceptable, or 4) continue testing to more precisely defined expected impact. The overall approach is to screen for gross effects of a contaminant such as acute toxicity and estimated chronic toxicity in the earlier tiers. At the higher tiers, the focus is on conducting tests for the more subtle effects of the contaminant such as impairment of reproduction. These higher tier tests only need be conducted if the lower tiers suggest that there is a potential for this type of water quality problem. As shown in Figure 4, with each succeeding tier of testing a more precise estimate of both the actual expected environmental concentration and also the "no impact" concentration for organisms is obtained.

The tiered environmental hazard assessment approach will likely be of particular significance in developing water quality management plans for urban lakes. It is likely that the results of a few screening bioassays normally associated with lower tiers of testing will show that urban stormwater drainage does not contain sufficient amounts of contaminants to cause toxicity. Findings of this type can greatly simplify the study program and thereby reduce its cost compared to the normal procedure for studying water quality involving routinely collecting data for a year or two and then attempting to interpret the data at the end of the study period in terms of the waterbody's water quality. Most water quality studies as conducted today provide limited amounts of information that can be used to formulate cost-effective, technically valid and at the same ecologically protective water quality management programs.

Control of Toxics

A key component of the hazard assessment approach for urban lakes as well as other waterbodies is the determination of the availability of the contaminants in the inputs to the lake. As discussed by Lee et al. (1979a), Jones and Lee (1978b), and Jones et al. (1979), chemical leaching tests for particulate matter such as are frequently used, do not in general properly assess the amounts of available forms of contaminants that can affect

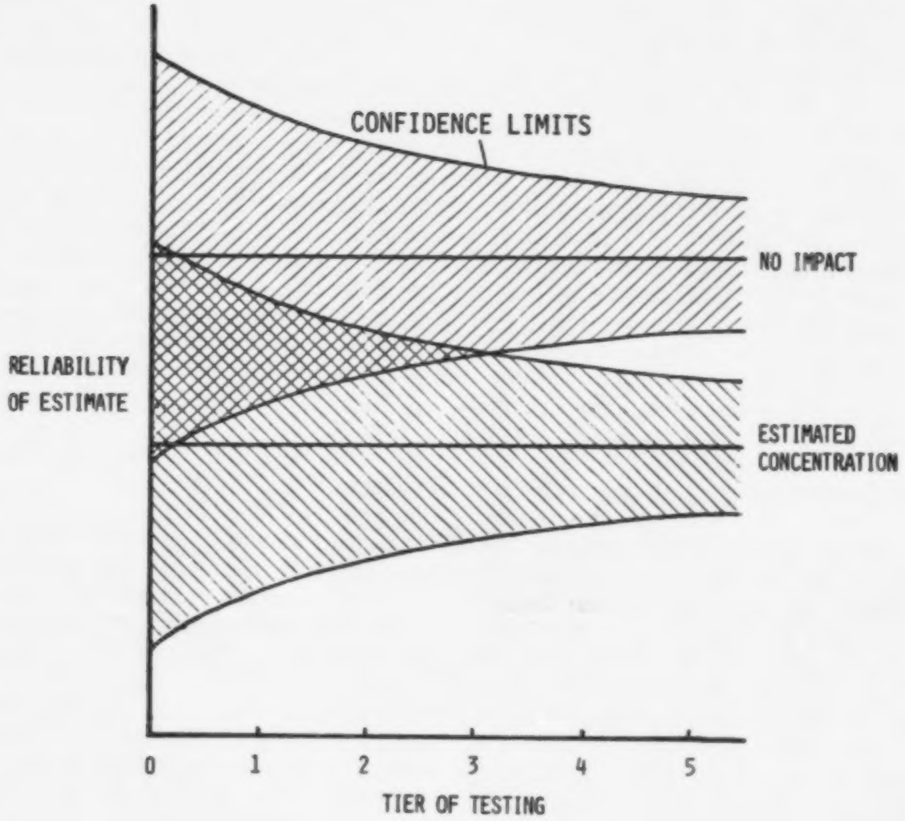


FIGURE 4 - Aquatic Toxicology and Environmental Chemistry-Fate Modeling Results for a Tiered Environmental Hazard Assessment

water quality for conditions typically found in urban lakes. Instead, bioassay procedures must be used on the sources of contaminants because of the variety of contaminants in the sources and for urban lakes, the typically high suspended particulate levels. If chemical leaching tests are used, the results must be verified against an extensive series of bioassays conducted at various times of the year in order to determine if there are any seasonal effects on the availability of the contaminants.

The first tier of an environmental hazard assessment for toxicants associated with contaminant inputs to urban lakes should be a 96 hour static bioassay screening test of the type described by Lee *et al*. (1978a) and Mariani (1979), on the undiluted water from each input. If no acute toxicity is found in these waters, and if there is a population of warm water pan fish (i.e., bluegill, sunfish, etc.) which are reproducing in the waterbody, and if there is at least a 100-fold dilution of the input waters with the lake water within a few days to a week or so of input, then the likelihood of there being any type of toxicity, including impairment of reproduction due to this source, is remote. The 100-fold dilution is based on the factor that is typically found to relate acute lethal toxicity to the chronic safe limit.

If acute toxicity is found in the screening bioassys of the undiluted urban stormwater drainage or other source of contaminant for an urban lake, then additional bioassays must be conducted on diluted contaminant source water, using appropriate dilutions with lake water based on the characteristics of the waterbody, to check for both acute and chronic toxicity. If potentially significant toxicity is found, it would be important to attempt to identify the specific toxic component. This can possibly be accomplished using the standard additions approach described by Lee and Jones (1979d). For further information on the use of bioassay procedures in a hazard assessment, consult Lee *et al*. (1979b).

Since bioaccumulation of contaminants in fish is of concern in urban lakes, it is imperative that some work also be done in the early tiers of any hazard assessment program to determine whether excessive buildup of contaminants is occurring in organisms used for human food. In general, the only criteria by which one can judge the significance of the body burden of some contaminants within an aquatic organism is the Food and Drug Administration (FDA) Action Limit. Therefore the most appropriate and only reliable method for assessing potential water quality problems associated with certain contaminants is to collect from the lake of concern seasonally over at least one year, representative samples of fish being used as human food and analyze their flesh for each of those contaminants for which the FDA has established Action Limits such as PCB's, mercury, DDT and its analogs, as well as several other chlorinated hydrocarbon pesticides.

If the concentrations of any of those contaminants would cause the fish to be unsuitable for use as food, i.e., the concentrations are above the FDA Action Limits, then each source of that contaminant must be investigated and control programs initiated where possible.

It is important to note that for developing control programs, while first priority must be given to analyzing fish flesh for those chemicals for which Action Limits exist since those are the only interpretable data at this time, as funds permit, the fish should be analyzed for any other contaminant which may be present in large amounts or may be of concern for a particular system. While these data may not at this time be interpretable in terms of water quality and fish usability, they may be a clue to potential water quality problems that should be investigated further since organisms in aquatic systems tend to be integrators for some types of persistent contaminants.

Control of Eutrophication

While it is expected that toxicants present in urban stormwater drainage and other urban lake water sources would not in general have a significant adverse effect on urban lake water quality, nutrients (nitrogen and phosphorus) frequently cause significant water quality deterioration in these waters. The hazard assessment approach for evaluating the significance of nutrient inputs to waterbodies, including urban lakes, is based on the OECD eutrophication modeling approach. The Organization for Economic Cooperation and Development (OECD) has sponsored a 22 country, 200 waterbody (lake and impoundment) study of the relationships between the nutrient load to a waterbody and the eutrophication response of that waterbody. Based on concepts originally developed by Vollenweider (1968) and the data on the approximately 40 OECD waterbodies in the U.S., Rast and Lee (1978) found that good correlations existed between the P load to a waterbody normalized by the waterbody's mean depth and hydraulic residence time, and the average summer planktonic algal chlorophyll concentrations, planktonic algal-related water clarity as measured by average summer Secchi depth, and for those waterbodies which thermally stratify, hypolimnetic oxygen depletion rate. Figure 5 shows these relationships. Included in this data set were several urban lakes including Lake Wingra in Madison, Wisconsin. While some have claimed that the nutrient behavior in small shallow lakes of this type is not the same as that in larger, deep waterbodies, the OECD eutrophication model has demonstrated that when normalized by hydrologic and morphologic characteristics, small and large lakes and impoundments behave in a similar manner in terms of eutrophication response to nutrient inputs. Lee et al. (1977, 1978b) have presented summaries of these results for the US OECD waterbodies.

Subsequent to the completion of the Rast and Lee (1978)

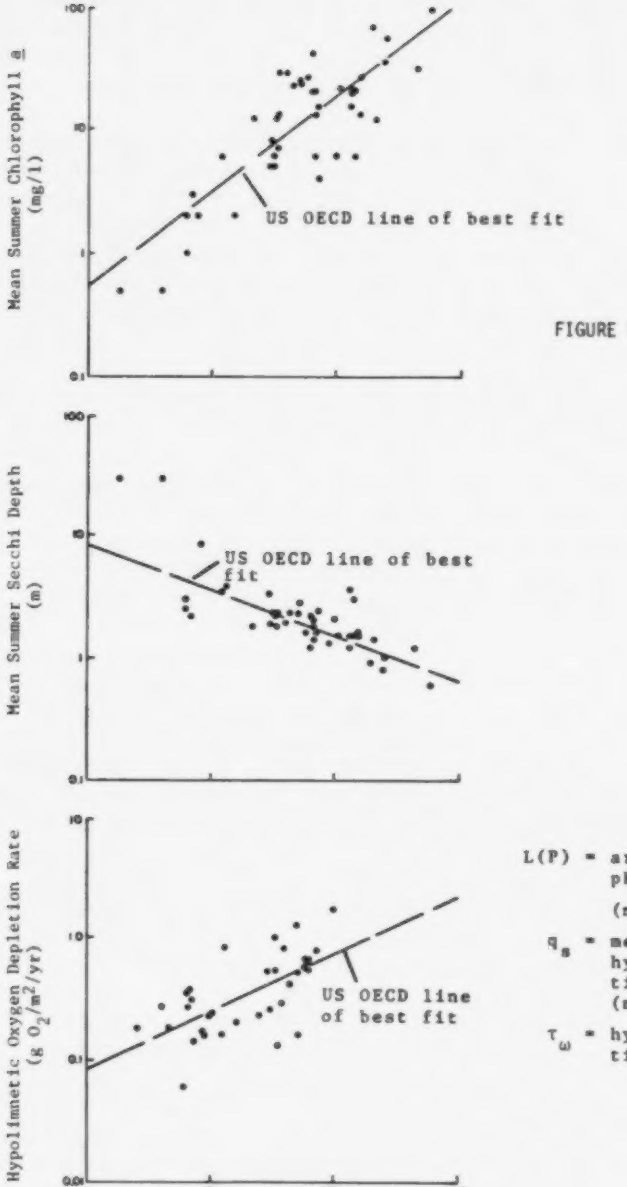


FIGURE 5 - US OECD Data Applied to Phosphorus Loading - Mean Chlorophyll a - Seechi Depth, and - Hypolimnetic Oxygen Depletion Rate Relationships

KEY

$L(P)$ = areal annual phosphorus load ($mg\ P/m^2/yr$)

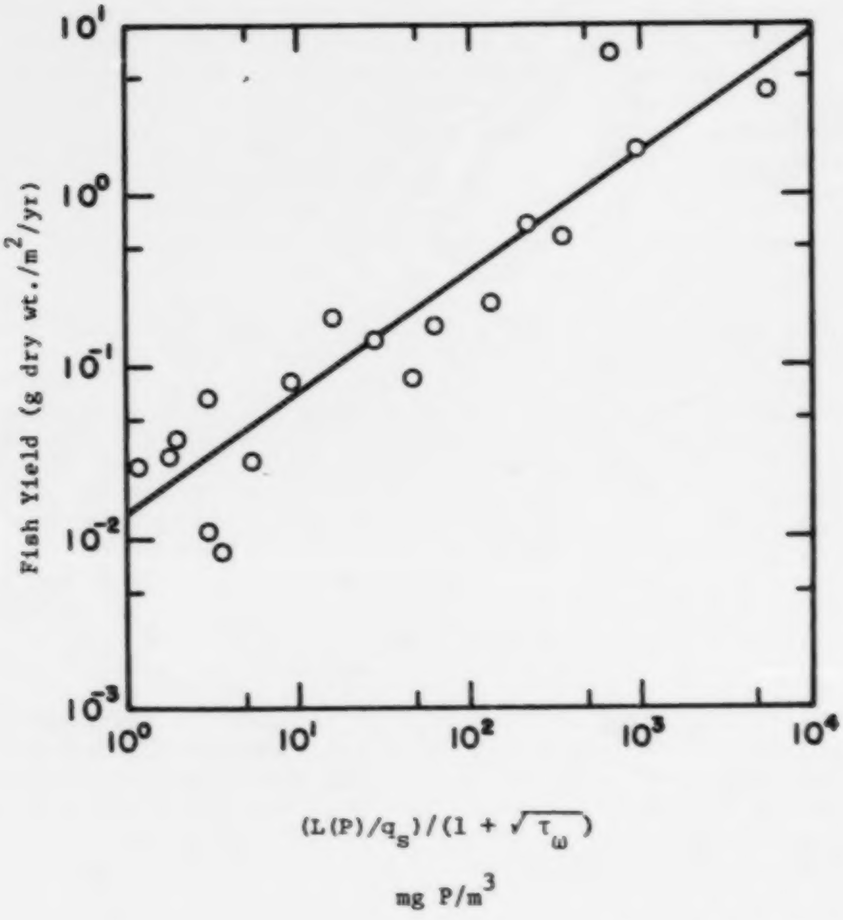
q_s = mean depth ÷ hydraulic residence time (m/yr)

τ_ω = hydraulic residence time (yr) work, Lee and his associates have evaluated the nutrient load-response relationships for an additional 60 or so waterbodies primarily in the U.S. and have found that these waterbodies follow the same relationships shown in Figure 5. Further, the other approximately 150 OECD waterbodies' nutrient load-eutrophication response relationships also are in agreement with what Rast and Lee (1978) found for the US OECD waterbodies.

Lee and Jones (1979c), who have conducted an extensive review of the effect of eutrophication on fisheries, have used the data provided by Oglesby (1977) and Rast and Lee (1978) to develop the relationship between normalized P load and fish yield shown in Figure 6. As would be expected, it shows that the greater the nutrient load, the greater the overall fish yield. Lee and Jones (1979c) should be consulted for further information on this topic.

There are a number of constraints on the use of the OECD eutrophication model that must be considered for each waterbody to which the approach is to be applied. While studies are being conducted on application of this approach to N-limited waterbodies, currently this approach is only applicable to those waterbodies whose maximum summer phytoplankton biomass is limited by phosphorus. The limiting nutrient in a waterbody or an area of a waterbody can be determined by measuring the concentrations of available N (NO_3^- plus NH_3) and available P (soluble ortho P) during the period of maximum phytoplankton biomass. In general, if the available P concentration is reduced to a few μg P/l, the phytoplankton growth at the time the samples were collected was most likely limited by P. If the available N concentrations are reduced to about 30 to 50 μg/l or so, N is likely limiting phytoplankton production. If neither nutrient is reduced to these levels, some other factor is likely limiting maximum planktonic algal biomass.

The ratio of available N to available P is also used to indicate which nutrient would be depleted first (i.e., potentially limiting) in a water, based on the theoretical uptake ratio of these nutrients by algae of 7.5 N to 1 P on a mg/l basis. Algal assays are also used to estimate the limiting nutrient by determining which nutrient if added would promote increased algal growth. Caution must be exercised in using the latter two approaches to determine the limiting nutrient; they must be performed near the time of maximum algal production since the results of such tests run at other times of the year will not necessarily give an accurate representation of the limiting nutrient during the period of water quality concern. Further, analyses for available N and P during peak biomass production should be conducted in conjunction with these procedures to verify that one of these nutrients is actually limiting the growth; while an N-to-P ratio may indicate a lesser relative abundance of one nutrient, some other factor such as light may in reality



Line of best fit:

$$\text{Fish Yield} = 0.7 \log [(L(P)/q_s)/(1 + \sqrt{\tau_\omega})] - 1.86$$

$$(r^2 = 0.86)$$

FIGURE 6 - Relationship between P load and Fish Yield

limit algal growth during the period of concern.

The OECD eutrophication modeling approach is also only applicable to waterbodies in which aquatic plant growth is dominated by phytoplankton. In its current degree of development it cannot be used to assess eutrophication response as measured by a macrophyte or attached algal growth parameter. As discussed by Newbry et al. (1979), it is likely possible to develop nutrient load-response relationships for macrophytes where response is assessed in terms of the percent of the area of the waterbody with depth less than 2m that is covered by aquatic macrophytes and attached algae.

Another constraint on the application of the OECD eutrophication model is that the average hydraulic residence time (i.e., filling time - volume divided by annual input) of the waterbody must be two weeks or more. For waterbodies having an annual hydraulic residence time shorter than two weeks, it may be possible to modify the model as was done by Jones and Lee (1978a) for Lake Lillinonah, CT. For this waterbody the summer average hydraulic residence time was used since each spring, the waterbody is essentially completely flushed.

WATER QUALITY MANAGEMENT IN URBAN LAKES

The overall approach that should be used for management of water quality on urban lakes is the same as for any waterbody, namely, limiting the input of available forms of contaminants to the extent necessary to achieve the desired water quality. In assessing available forms, it is important to assess not only those forms which are immediately available but also those which can become available in the waterbody. Because of the potential significance of urban stormwater drainage as a source of contaminants for urban lakes, in those situations where the hazard assessment shows that this source is a significant source of contaminants causing water quality deterioration, studies should be done to define the source of contaminants in the urban stormwater drainage. At this time there is a relatively poor understanding of the specific sources of total as well as available forms of contaminants in this source. The studies of Cowen and Lee (1973) and Kluesener and Lee (1974) have shown that in the fall, an appreciable part of the P present in urban stormwater drainage is from fallen leaves. Lee (1972) proposed that the rapid removal of leaves and frequent street sweeping using vacuuming could significantly reduce the phosphorus content of urban stormwater drainage. Ahern and Armstrong (1979) found that street sweeping reduced the P content of urban stormwater drainage by 47 to 59%.

The importance of focusing nutrient control programs on available forms of nitrogen and phosphorus was demonstrated by the studies of Lee and his associates in Madison, WI. Kluesener and Lee (1974) found that urban stormwater drainage contributed about 0.1 $g/m^2/yr$ of total phosphorus and 0.5 $g/m^2/yr$ of total nitrogen. About 25% of the total N was in the form of ammonia and nitrate, while about 50% of the total P was in the form of soluble ortho P. Cowen et al. (1978), using bioassay techniques found that about 50% of the particulate organic and inorganic N (i.e., the difference between the total N and the sum of nitrate and ammonia) would likely become available to stimulate algal growth in an urban lake. They also found that only about 20% of the difference between the total P and soluble ortho P would likely become available to support algal growth. These results point to the fact that appreciable parts of the N and P entering urban lakes from urban stormwater drainage are in forms not likely to support algal growth in the waterbody. While the results of the studies by Cowen et al. (1978) are in general agreement with what other investigators who have conducted similar studies have found, additional work needs to be conducted for each system of interest until sufficient information is gathered to make appropriate estimates of nutrient export from urban areas and the availability of these nutrients to support algal growth.

A potentially significant source of nutrients for some urban lakes is waterfowl. While some of these birds only alter recycle of nutrients within the waterbody, such as ducks which feed on aquatic organisms, other waterfowl such as geese feed on land but spend most of their time in the water. The latter type of bird can add nutrients from on-land sources which may never otherwise reach the lake. An example of the potential importance of this problem is seen in urban lakes in the City of Fort Collins, Colorado, where the population of resident geese number well into the tens of thousands.

Typical urban stormwater and/or combined sewer contaminant control programs are usually directed toward the control of particulate forms of contaminants. Most of these programs are based on the erroneous premise that there is a relationship between the total load of a contaminant and its impact on water quality. As discussed above, it is rare that the total load or concentration of a contaminant can be used to predict the impact of the contaminant on water quality. While control of particulate forms of a contaminant may be useful for siltation control, it may not be cost-effective for reducing the magnitude of chemical related water quality problems. In many instances the most cost-effective way to deal with siltation, is periodic dredging.

The first priority in any urban lake's water quality maintenance program should be the removal of debris-trash usually left by the users of the area. The next priority should be given to the control of excessive fertility. Obviously, all significant point sources of nitrogen and/or phosphorus depending on what is limiting planktonic algal growth in the waterbody, should be identified. If the hazard assessment study shows P is limiting or can be made limiting through point source control, and the point source(s) already has some treatment for removal of solids and BOD, then P removal by alum or iron co-precipitation techniques should be added to the treatment process. Generally, P can be controlled in domestic wastewater to less than 1 mg P/l total P at a cost of less than a half a cent (0.5¢) per person per day for the population served. As noted above, the OECD eutrophication study modeling results can be used to estimate the magnitude of planktonic algal-related water quality improvement associated with point source and diffuse source nutrient control. Before embarking on large-scale street vacuuming or other diffuse source control programs, the magnitude of water quality improvement that could be achieved by these measures should be determined using the OECD eutrophication model. Such control programs are often relatively expensive and frequently provide limited reductions in available nutrient loads. As shown in Figure 5, relatively large reductions in P load are needed to significantly change the planktonic algal-related chlorophyll concentration. This is especially true for eutrophic lakes in which even relatively large reductions in chlorophyll may not result in a perceivable improvement in water quality.

As a result of their shallow character, many urban lakes have significant water quality problems due to attached algae and macrophytes. As discussed by Lee (1973) and Lee and Jones (1979c), a variety of techniques such as aquatic herbicides, harvesting, etc., are available for control of these growths. In general, the overall control philosophy for aquatic macrophytes and attached algae should be to remove the fewest number of these plants as necessary to make the waterbody usable for the desired recreational purposes. As discussed by Lee and Jones (1979c), aquatic macrophytes and especially attached algae compete with planktonic algae for available nutrients. Extensive removal of attached algae and macrophytes can result in production of phytoplankton blooms and could destroy fish habitat and nursery grounds.

The dredging of urban lakes for the removal of deposited sediment can also have a significant impact on the relative utilization of nutrients by various types of aquatic plants. Extensive dredging to depths of 3m or more can greatly decrease the macrophyte and attached algae in eutrophic lakes with a concomitant increase in phytoplankton. Dredging should be restricted to those parts of the waterbody for which the shallowness permits excessive growths of attached algae and macrophytes which significantly interfere with recreational uses of the water such as swimming, boating, or fishing.

The direct alum addition to urban lakes is a technique that deserves much greater attention for managing urban lake water quality than it has received in the past. In many situations it may be the only technique available to manage excessive fertilization problems. Since urban lakes tend to normally be turbid, alum flocculation of inflowing waters can help reduce turbidity and also remove P from the watercolumn. The alum floc that forms can be settled either directly in the lake or in a pre-impoundment constructed for this purpose. Systems can be developed to automatically feed alum to the stormwater, combined sewer overflow, or domestic wastewater bypass inputs to the lake. This approach does not appear to be harmful to fish as long as there is adequate alkalinity in the water to maintain a desirable pH. For low alkalinity waters, it may be necessary to add lime to increase the buffer capacity of the water. The duration of effectiveness of this approach is directly dependent on the hydraulic residence time of the waterbody. Waterbodies with several years' residence times will require relatively infrequent treatment. However, the typical urban lake would require the more or less continuous treatment of all inflowing waters to be effective. It should be noted that such treatment may result in increased amounts of attached algae and macrophyte growth because of the greater water clarity arising from alum treatment. This may necessitate additional harvesting or other control programs especially for macrophytes which can derive at least part of their nutrients from the sediments.

Those responsible for developing nutrient control programs for waterbodies are frequently concerned about the rate of recovery of the waterbody upon altering the nutrient load. The shallowness of many urban lakes has led some to believe that these waterbodies will not recover upon reduction of nutrient load because of the release of nutrients from the sediment. As discussed by Sonzogni *et al*. (1976), the rate of recovery of a waterbody upon reduction of P input depends on the P residence time of the waterbody. About 95% of the expected recovery from the degree of altered load occurs within a period equal to three times the phosphorus residence time. The P residence time of many waterbodies is less than one year which means that within a few years after altering the load, a new equilibrium P-chlorophyll concentration will be achieved for the waterbody.

CONCLUSIONS

Increasing pressures are being placed on water pollution control agencies to manage water quality in urban lakes. The first step in developing a water quality management plan for such a waterbody is a careful definition of the water quality problems that may exist therein. In this evaluation, for many urban lakes, particular attention will have to be given to eutrophication-related water quality which will require reduction in the amounts of available forms of nitrogen and phosphorus entering them. The

OECD eutrophication study results should be used to describe the nutrient load-response relationships for the urban lakes and most importantly, should be used to estimate the magnitude of water quality improvement that will arise as a result of altering the nutrient loads to P limited urban lakes. In many instances it may not be possible to control nitrogen and/or phosphorus to a sufficient degree to attain desired water quality in urban lakes because of the difficulty of limiting nutrient input from urban stormwater drainage and/or combined sewer overflow. Direct alum addition to the inflowing waters is a technique that should be more thoroughly investigated as a tool to manage excessive fertility in urban lakes. Also selective dredging and attached algae and macrophyte control through harvesting can be effective tools for minimizing impact of aquatic plant nutrients on urban lake water quality.

For those waterbodies impacted by toxicants as well as nutrients, an environmental hazard assessment should be conducted to define the magnitude of the impact of contaminant sources on water quality, i.e., beneficial uses, of the lake. Such an assessment can also be used to develop the information necessary to control the excessive input of the contaminants. Water quality standards adapted directly from the US EPA Red Book criteria have limited applicability in serving as a basis for developing water quality management programs for urban lakes. The environmental hazard assessment involving the tiered development of environmental chemistry-fate and aquatic toxicology information with emphasis given to use of bioassays to define available forms of the contaminant should be used.

REFERENCES

Ahern, J. and Armstrong, D. E. "Phosphorus Control in Urban Runoff by Streetsweeping," Personal communication to G. Fred Lee, Civil Engineering, Colorado State University, Fort Collins, CO (1979).

Cairns, J., Dickson, K. L., and Maki, A. W. (eds.) Estimating the Hazard of Chemical Substances to Aquatic Life, American Society for Testing and Materials Special Publication 657, ASTM, Philadelphia, PA (1978).

Cowen, W. and Lee, G. F. "Leaves as a Source of Phosphorus," Environ. Sci. and Tech. 7:853-854 (1973).

Cowen, W. F., Sirisinha, K., and Lee, G. F. "Nitrogen and Phosphorus in Lake Ontario Tributary Waters," Water, Air, and Soil Pollution 10:343-350 (1978).

Jones, R. A. and Lee, G. F. "Evaluation of the Impact of Phosphorus Removal at the Danbury, Connecticut Sewage Treatment Plant on Water Quality in Lake Lillinonah," Environmental Engineering, Colorado State University, Fort Collins, Occasional Paper No. 31, April (1978a).

Jones, R. A. and Lee, G. F. "Evaluation of the Elutriate Test as a Method of Predicting Contaminant Release during Open Water Disposal of Dredged Sediment and Environmental Impact of Open Water Dredged Material Disposal, Vol. I: Discussion," Technical Report D-78-45, U.S. Army Corps of Engineers WES, Vicksburg, MS (1978b).

Jones, R. A., Mariani, G. M., and Lee, G. F. "Evaluation of the Significance of Sediment-Associated Contaminants to Water Quality," Presented at American Water Resources Assoc. Conference, Las Vegas, NV (1979). To be published in conference proceedings.

Kluesener, J. W. and Lee, G. F. "Nutrient Loading from a Separate Storm Sewer in Madison, Wisconsin," Water Poll. Cont. Fed. 46:920-936 (1974).

Lee, G. F. "Ways in Which a Resident of the Madison Lakes' Watershed May Help to Improve Water Quality in the Lakes," A report of the Water Chemistry Program, University of Wisconsin (1972).

Lee, G. F. "Eutrophication," Transactions of the Northeast Fish and Wildlife Conference, pp 39-60 (1973).

Lee, G. F. and Jones, R. A. "The Role of Biotransformation in Environmental Hazard Assessment," To be published in Proc. of 'Biotransformation and Fate of Chemicals in the Aquatic Environment' workshop held in Pellston, MI, August (1979a).

Lee, G. F. and Jones, R. A. "The Role of Environmental Chemistry-Fate Modeling in Environmental Hazard Assessment: An Overview," Presented at ASTM Symposium on Aquatic Toxicology, Chicago, IL, October (1979b). To be published in conference proceedings.

Lee, G. F. and Jones, R. A. "Effect of Eutrophication on Fisheries," Prepared for and to be published by the American Fisheries Society (1979c).

Lee, G. F. and Jones, R. A. "Water Quality Characteristics of the US Waters of Lake Ontario during the IFYGL and Modeling Contaminant Load - Water Quality Response Relationships in the Nearshore Waters of the Great Lakes," Report to NOAA, Ann Arbor, MI, June (1979d).

Lee, G. F., Rast, W., and Jones, R. A. "Recent Advances in Assessing Aquatic Plant Nutrient Load-Eutrophication Response for Lakes and Impoundments," Environmental Engineering, Colorado State University, Fort Collins, Occasional Paper No. 14, May (1977).

Lee, G. F., Jones, R. A., Saleh, F. Y., Mariani, G. M., Homer, D. H., Butler, J. S., and Bandyopadhyay, P. "Evaluation of the Elutriate Test as a Method of Predicting Contaminant Release during Open Water Disposal of Dredged Sediment and Environmental Impact of Open Water Dredged Materials Disposal, Vol. II: Data Report," Technical Report D-78-45, U.S. Army Corps of Engineers WES, Vicksburg, MS, 1186 pp, August (1978a).

Lee, G. F., Rast, W., and Jones, R. A. "Eutrophication of Waterbodies: Insights for an Age-old Problem," Environ. Sci. & Technol. 12:900-908 (1978b).

Lee, G. F., Jones, R. A., and Rast, W. "Availability of Phosphorus to Phytoplankton and Its Implications for Phosphorus Management Strategies," Presented at IJC-Cornell University Conference on Phosphorus Management Strategies for the Great Lakes, April (1979a). To be published in conference proceedings.

Lee, G. F., Jones, R. A., and Newbry, B. W. "Use of Bioassays in a Hazard Assessment," Presented at American Water Resources Assoc. Conference, Las Vegas, NV (1979b). To be published in conference proceedings.

Mariani, G. M. Bioassay Assessment of Toxicity of Contaminants in Dredged Sediment, Ph.D. dissertation, Univ. of Texas at Dallas, Richardson, TX (1979).

Newbry, B. W., Jones, R. A., and Lee, G. F. "Application of the OECD Eutrophication Modeling Approach to Cherokee Reservoir and Other TVA Impoundments," Report to the Tennessee Valley Authority, Chattanooga, TN (1979).

Oglesby, R. T. "Relationships of Fish Yield to Lake Phytoplankton Standing Crop, Production, and Morphoedaphic Factors," Journal of the Fisheries Research Board of Canada, 34:2271-2279 (1977).

Rast, W. and Lee, G. F. "Summary Analysis of the North American (U.S. Portion) OECD Eutrophication Project: Nutrient Loading-Lake Response Relationships and Trophic State Indices." US EPA, EPA-600/3-78-008, Corvallis, OR, 454 pp (1978).

Sonzogni, W. C., Uttormark, P. C., and Lee, G. F. "A Phosphorus Residence Time Model: Theory and Application," Water Research 10:429-435 (1976).

US EPA, Quality Criteria for Water, EPA-440/9-76-023, U.S. Gov't. Printing Office, Washington, D.C. (1976).

Vollenweider, R. A. "Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication." Technical Report DAS/CSI/68, OECD, Paris (1968).

THE EFFECT OF URBAN STORMWATER RUNOFF ON THE WATER QUALITY OF LAKE JACKSON, FLORIDA

Christian Byrne, Charles R. Donahue, William C. Burnett*
Department of Oceanography, The Florida State University
Tallahassee, Florida 32306

ABSTRACT

Analyses of the runoff from the southern end (Meginniss Arm) of Lake Jackson revealed high concentrations of suspended solids, dissolved nutrients , heavy metals particulates, and petro-chemicals. It is clear that the overwhelming majority of pollutants in this end of the lake is due to urban stormwater runoff. In response to the serious water quality in the southern areas of Lake Jackson, the Florida Department of Environmental Regulations in association with the United States Environmental Protection Agency have proposed to construct and maintain a bio-filtration system in the watershed of Meginniss Arm to restrict the urban pollutional loading.

The primary source of petroleum hydrocarbon in Meginniss Arm is the waste automobile oil from the roads and parking areas of the commercial establishments of the area. Lake Jackson offords an area for a very interesting study of pollutant loading. There are two watersheds in Lake Jackson, which are similar in size, topography, and geology but very dissimilar in land usuage. Ox-Bottom Creek Arm in the Northern area is primarily forested-agricultural with little mechanical activity while Meginniss Arm, is highly urban containing two shopping malls. These major watersheds will make possible a comparison of the concentration of hydrocarbons in the stormwater runoff from the two areas and make possible the separation of the contribution of biogenic and anthropogenic sources in the two areas. By sampling the stormwater runoff (for dissolved and particulate forms), atmospheric dust, rainfall, sediments, and water samples from several locations of Lake Jackson, it might be possible to determine a mass balance budget for hydrocarbons in Lake Jackson, Florida.

INTRODUCTION

Lake Jackson, in Leon County Florida (Figure 1), is a closed depression (1960 ha.) in an area of rapidly changing land usage. Urban development has increased markedly in the southern watershed of Lake Jackson, while the northern areas have remained essentially undisturbed with little or no development occurring. Intensive studies of the lake and its associated watershed (Smith, 1972; Smith, 1973; Smith, 1974; Schamel _et al_., 1974; Harris and Turner, 1974, Turner _et al_., 1974) have revealed significantly

---

*Department of Oceanography, The Florida State University, Tallahassee, Florida 32306

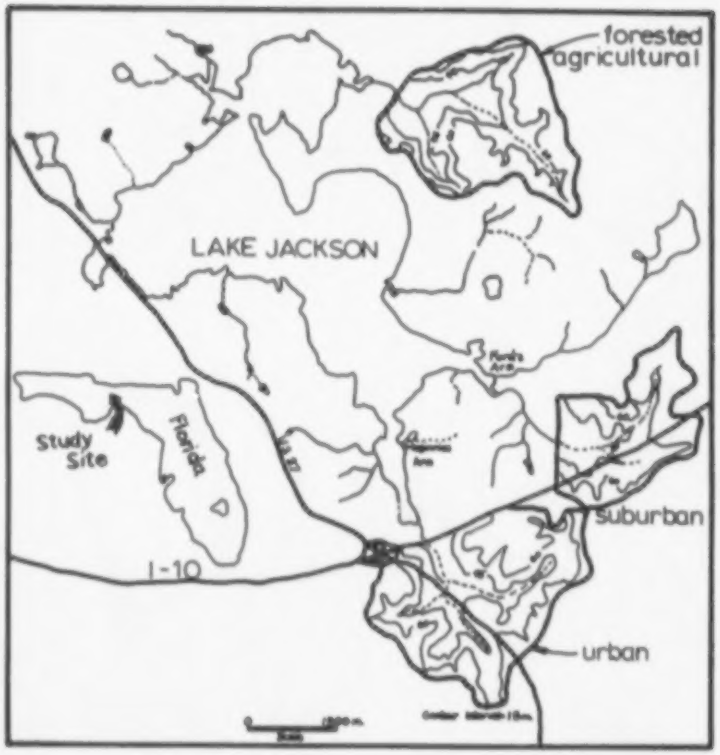


Figure 1. Lake Jackson and its surrounding watersheds.

poorer water quality in the southern areas. This decrease in water quality can be directly related to the stormwater runoff from the urbanized watersheds of these areas.

The watershed possessing the worst water quality is also the one with the highest degree of urban development - the watershed of Meginniss Arm. Intensive sampling and analysis of stormwater runoff in the area (Harriss and Turner, 1974) demonstrated the runoff to be a principal source of organic solids, dissolved nutrients, and particulate forms of several heavy metals, including lead. Recent investigations in other areas (Wakeham, 1977; Hunter et al., 1979) have shown that stormwater runoff can be a significant source of organic compounds and, in particular, petroleum hydrocarbons. The object of this study was to determine the presence of petroleum hydrocarbons in the stormwater runoff to Meginniss Arm during a storm event and assess the concentrations and forms of these hydrocarbons as they move through the discharge channel and empty into Lake Jackson.

STUDY SITE

The study site was the urban watershed of Meginniss Arm (Figure 2). A case study was made of the stormwater runoff generated in the watershed during a storm event on November 27, 1978. Table 1 indicates the general land usage and hydrologic characterization of the storm event. Water grab samples were taken at four different locations along the discharge tributary. The first sampling station was located below the discharge conduit at 24.8 ha. The second sampling station was located below Tallahassee Mall (TM) discharge conduit system. The drainage area was 202.9 ha. The third station was at the entrance to a series of small holding ponds (P) above Interstate 10. This station received the joint discharge from the Tallahassee Mall and Northwood Mall sheds. This total drainage was 356.9 ha. The final station was located at the mouth of the discharge channel as it emptied into Meginniss Arm at the southern portion of Lake Jackson.

Sampling times were previously determined from a study of twelve hydrographs of storm events in this watershed (Harriss and Turner, 1974). Samples were taken at the initial discharge of runoff from the storm (time = 0 hours), 0.5 hours, 1 hour, 2 hours, 4 hours and 8 hours after the initial discharge. Only at the fourth station (MA) was this series changed, the first sample was taken at 0.5 hour and the last 26 hours after the initial discharge.

METHODS AND MATERIALS

Grab samples were taken at the selected time intervals using steel cans (Firestone Challenger VI). These cans were prewashed with warm water and Alconox, rinsed, then washed with single distilled methanol, and finally single distilled dichloromethane. All rubber seals were wrapped with teflon film to prevent contamination.

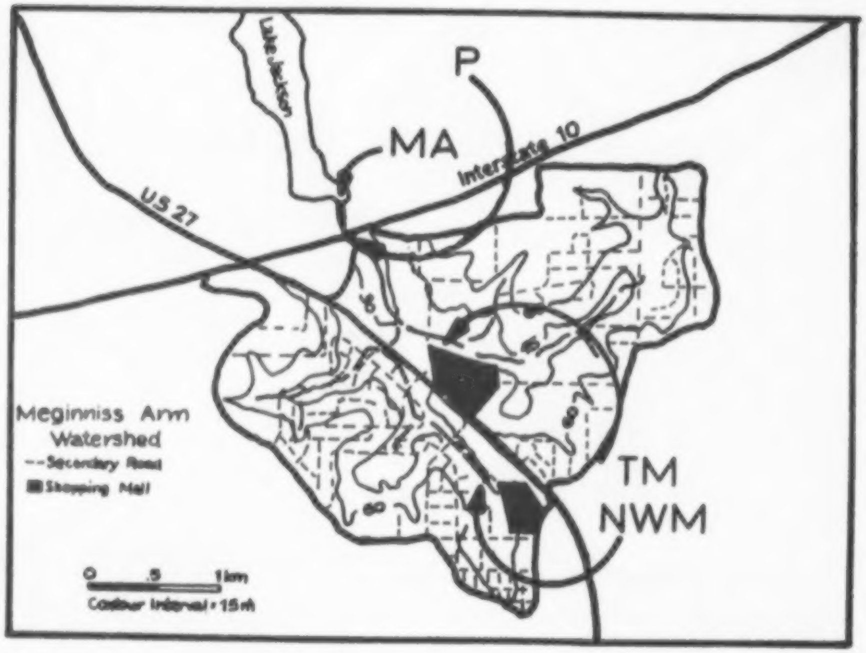


Figure 2. The urban watershed of Meginniss Arm

TABLE 1. Case study of stormwater runoff generated in Megginnis Arm on November 27, 1978. Rainfall and discharge data supplied by the North Florida District Office of the U.S. Geological Survey.

Land Use Characterization (Harris and Turner, 1974)

Type	Area	Percent
Woodlands	81.3 ha.	22.8
Agricultural	19.8 ha.	5.6
Residential	195.9 ha.	54.9
Commercial	45.7 ha.	12.8
Construction	14.2 ha.	3.9
	356.9 ha.	100.0
Impervious Area (roads, parking, roofs)	91.5 ha.	25.6

Hydrologic Characterization

Total Precipitation	2.51 cm (92.4 X 10^6l)
Gauged Runoff at Pond	0.31 cm (26.9 X 10^6l)
Water Yield	29.1 %
Antecedent Conditions	No significant precipitation previous 20 days

The water grab samples were separated into two fractions (dissolved and particulate) and prepared for analysis as follows:

Dissolved Fraction

The dissolved fractions were filtered through pre-combusted (500°C; 12 hours) GF/F glass filter filters (142 mm) under pre-purified nitrogen gas pressure. The dissolved fraction was transferred into separatory funnels and acidified below 2 pH and extracted three times with dichloromethane at a ratio of 10:1. This total organic extract (TOE) was dried over anhydrous $MgSO_4$, reduced to dryness and weighed on a MicroMettler M5 balance to the nearest microgram.

Particulate Fraction

The particulate fractions were trapped on pre-combusted GF/F filters (effective efficiency - 1.0 um). The filters were then folded, placed in aluminum foil, and frozen at - 80°C until extraction. The filters were thawed at the time of extraction, cut into 0.5 cm stips, and placed into glass thimbles for Soxhlet extraction. The filters (particulates) were extracted initially with methanol for 12 hours and then 12 hours with dichloromethane. The methanol extracts were back extracted twice with hexane. The hexane and dichloromethane extracts were combined, dried over anhydrous $MgSO_4$ and reduced to dryness under a vacuum. These extracts constituted the total organic extract (TOE). The TOE were dried and weighed as in the case of the dissolved fraction. From this point both fractions were treated identically.

All samples were saponified in round bottom flasks with a mixture of KOH/methanol: benzene: deionized water (Van Vleet and Quinn, 1977). After refluxing the solutions for at least four hours and then cooling, the samples were transferred to separatory funnels where the phases were separated by addition of deionized water. The non-saponificable extract (NSE) was extracted twice with benzene, dried and weighed. The NSE was then taken up in hexane for column chromatography. The NSE was applied to activated silica gel columns with a column to sample ratio of 200:1. The columns were previously charged with four columns volumes hexane. The columns were successively eluted with two column volumes of hexane to remove the aliphatic species and with two column volumes to remove the aromatic species. The two eluants were transferred to pre-weighed microvials, reduced, and weighed. The two eluants were then taken back up in their respective solvents.

The aliphatic eluants of the dissolved and particulate fractions were analyzed by gas-liquid chromatography (GLC) with a Hewlett-Packett Model 5711 gas chromatograph equipped with dual flame ionization detectors and dual 6 ft X 1/8 in. stainless steel columns containing 10% SP-1000 mesh Chromosorb W (HP) The columns were temperature programmed from 80° to 250° at a rate of 4°C/minute. The injection and detector temperatures were 250°. The nitrogen carrier gas flow rate was 51 ml/min. The hydrocarbons measured eluted from the columns between nC12-nC34. The chromatographs were examined for CPI (Carbon Preference Index) values and the presence of unresolved envelopes. Both are considered standard methods of assessing the presence of petroleum hydrocarbons (Blumer and Sass, 1972).

All solvents used were doubly distilled in all glass distillation systems unless otherwise stated. All glassware was washed with Alconox, rinsed, submerged in Non-Chromix-sulfuric acid solution for at least four hours, rinsed with water, air dried, rinsed with single distilled dichloromethane, and dried at 150°C overnight.

RESULTS

Concentrations of total dissolved hydrocarbons (TDH) and total particulate hydrocarbons (TPH) for the November, 1978 storm event are given in Table 2. These total concentrations were determined by the addition of the dry weights of the aliphatic and aromatic eluants of each fraction. Although dry weights of organic extracts often lose low boiling-low molecular weight compounds, gravimetric determinations of the concentrations of these extracts are considered more accurate than analysis of the gas-liquid chromatographs. Very high boiling-high molecular weight compounds do not elute from the chromatograph columns and as such would not be included in the final concentration values. In addition, the unresolved envelopes that occur in chromatographs can not be easily quantified by planimetry or computer assessment.

Dissolved Hydrocarbon Concentrations

An examination of Table 2 reveals that the TDH concentrations ranged from 19 to 72 ug/l (ppb). Although the range does not appear to be very great, there is a significant maximum which occurs two hours into the storm event. This could be the result of bleeding or leeching of dissolvable hydrocarbons from the impervious (i.e. asphalt) surfaces. This relationship is shown clearly in Figure 3. This figure presents data for the station at Tallahassee Mall (TM) during the storm event. There is a definite relationship between rainfall, discharge, and hydrocarbon concentrations.

Particulate Hydrocarbon Concentrations

The total particulate hydrocarbon (TDH) concentrations

TABLE 2. Concentrations of total dissolved hydrocarbons (TDH) and total particulate hydrocarbons (TPH). All concentrations are in units of ug/1.

Station	Time	Dissolved Aliphatic	Dissolved Aromatic	Total Dissolved (TDH)	Particulate Aliphatic	Particulate Aromatic	Total Particulate (TPH)
TM 1*	9:30	8	10	18	2456	7927	10383
TM 2	10:00	12	16	28	69	104	173
TM 3	10:30	9	12	21	25	19	44
TM 4	11:00	27	16	43	332	272	604
TM 5	13:20	22	9	31	83	58	141
TM 6	17:30	10	19	29	23	19	42
NW 1	9:35	42	29	71	15322	1043	16365
NW 2	10:05	24	32	56	714	581	1295
NW 3	10:35	7	17	24	462	298	760
NW 4	11:35	30	38	68	428	534	962
P 1	9:40	-	-	-**	1741	10831	12572
P 2	10:10	18	25	43	81	154	235
P 3	10:40	19	22	41	455	925	1380
P 4	11:40	29	24	53	131	241	372
P 5	13:35	30	9	39	317	310	672
P 6	17:40	16	14	30	29	8	37
MA 1	10:20	18	21	39	398	916	1314
MA 2	10:50	18	15	33	1043	38	1081
MA 3	11:50	13	14	27	116	411	527
MA 4	13:30	25	15	40	53	79	132
MA 5	17:50	24	22	46	72	76	148
MA 6	11:30 (11/28)	35	23	58	58	37	129

* Abbreviations used in this table: TM = Tallahassee Mall, NW = Northwood Mall, P = Pond, and MA = Meginniss Arm

** not determined

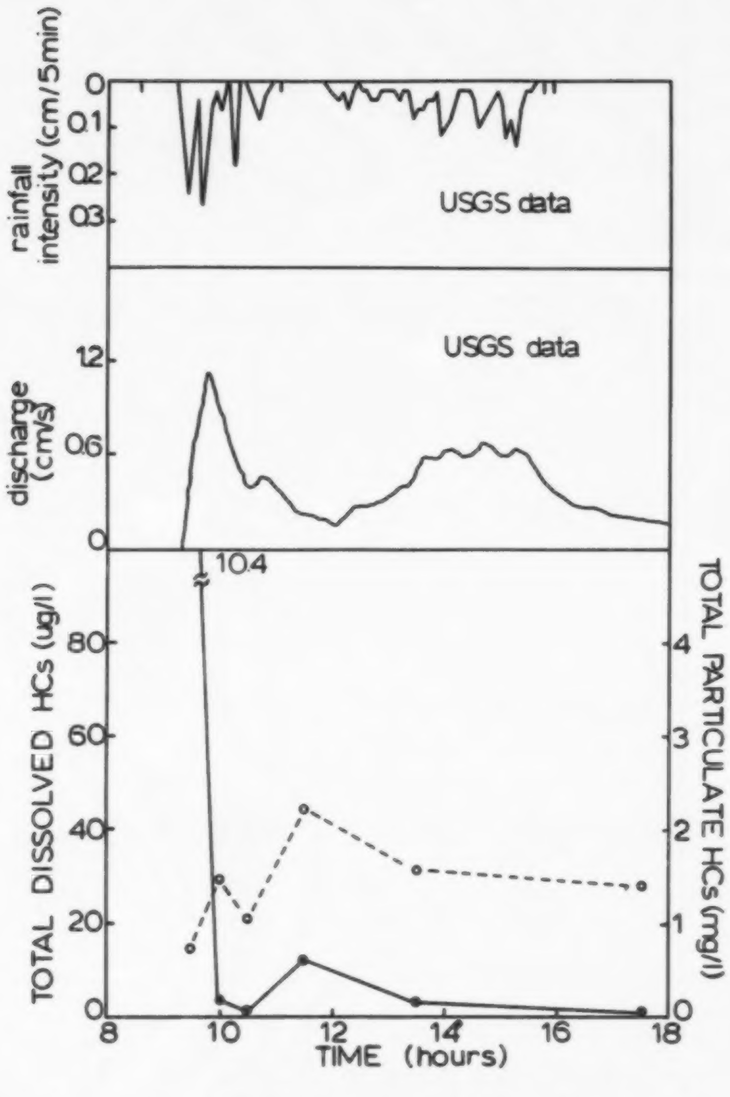


Figure 3. The relationship of rainfall intensity, discharge, and total particulate and dissolved hydrocarbon concentrations with time during the November 27, 1978 storm event.

were at least a factor of 2 greater than the total dissolved hydrocarbon (TDH) concentrations throughout the storm. In the initial runoff discharge at station NW#1, the difference was two orders of magnitude. Unlike the TDH concentrations the TPH concentrations reach their highest values at the beginning of the storm surge. After this surge had passed, the TPH mimicked the TDH concentrations with a secondary maximum at two hours into the storm at which point the concentrations began to plateau out very similarly to the TDH. The TPH as well as the TDH concentrations are both shown in Figure 3.

There appears to be no subsequent increase in either concentrations later into the storm with the additional discharge that occurs after 1300 hours. This may indicate that with continuous rinsing of the pavement no further hydrocarbons can be easily transported in the particulate or dissolved form. Analysis of the gas-liquid chromatographs of the aliphatic eluants of both the particulate and dissolved fractions reveal that petroleum hydrocarbons are definitely present in these fractions and that they are primary sources of the hydrocarbons present. Figure 4 shows chromatographs at various stages during the storm at the Tallahassee Mall station. The CPI (Carbon Prefernce Index) for the first chromatograph (Figure 4a) is approximately 0.99 CPI is a weighted measure of the ratio of odd carbon to even carbon hydrocarbons which can be useful in distinguishing natural from anthropogenic sources. Samples of natural biogenic hydrocarbons possess CPI's in a range from 1.5-8. Anthropogenic hydrocarbons (i.e. petroleum hydrocarbons) have CPI values approaching unity (1) or less. Figure 4b is a chromatograph of the aliphatic eluant of the TDH at Station TM two hours into the storm event. The CPI is 1.5 and, in addition, a unresolved hydrocarbon envelope, often indicative of petroleum hydrocarbons (Blumer and Sass, 1972), is present. This envelope, together with the CPI value, make a good case for the presence of petroleum hydrocarbons in this sample. The fact that this envelope appears after only two hours of flushing of the surface of the watershed is another factor that would support the theory of bleeding or leeching of the asphalt occurring into the stormwater runoff. A chromatograph of the aliphatic eluant of the TPHs from station TM taken at time 0 is shown in Figure 4c. The CPI value of 0.96 and the unresolved envelope give validity to our hypothesis that petroleum hydrocarbons are present in the particulate fraction of the stormwater runoff very early during the course of the storm.

DISCUSSION

Wakeham (1977) studied urban stormwater runoff which flows into Lake Washington from surrounding areas of Seattle, Washington. The values of total aliphatic hydrocarbons in the urban stormwater

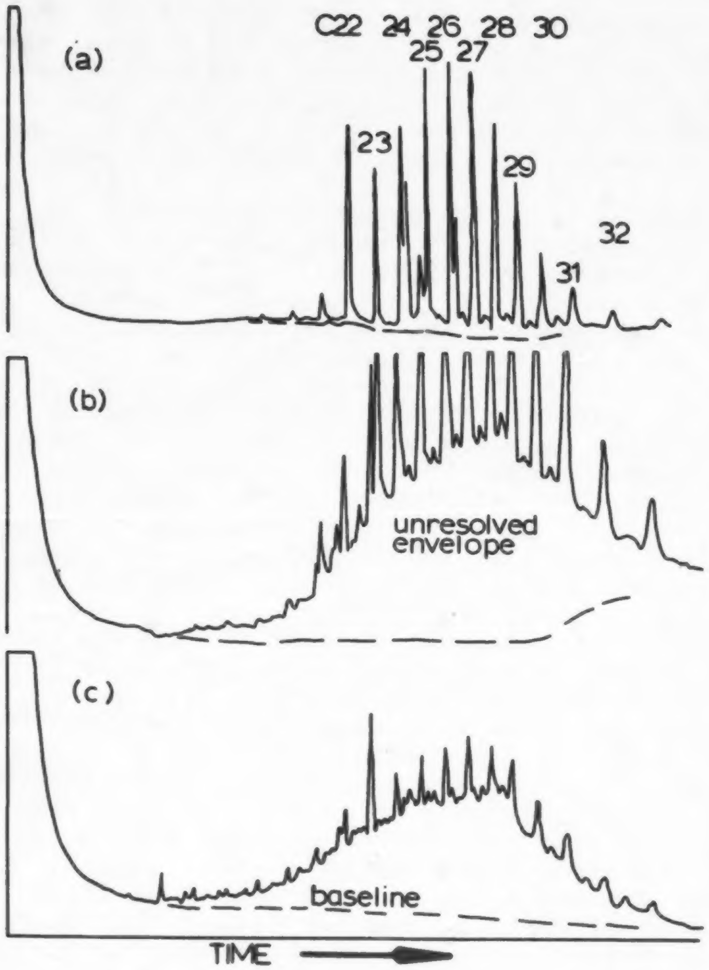


Figure 4. Gas-liquid chromatographs of
(a) Tallahassee Mall (9:30 - initial discharge surge); total dissolved aliphatic hydrocarbon fraction
(b) Tallahassee Mall (9:30 - initial discharge surge); total particulate aliphatic hydrocarbon fraction.
(c) Tallahassee Mall (11:30 - two hours after initial surge); total dissolved aliphatic hydrocarbon fraction.

runoff ranged from 200 to 7500 ug/l with a mean value of 1200 ug/l. These values were total aliphatic hydrocarbons as the particulate fraction was not separated from the dissolved. Wakeham's total aliphatic concentrations are comparable with the results of this study. The total aliphatic hydrocarbons concentrations reported here range from 36 to 15000 ug/l.

Hunter et al.(1979) studied five storm events and the associated urban stormwater runoff of a watershed of Philadelphia, Pennsylvania. The average total hydrocarbon concentrations for the five storms was 3.69 mg/l, with a total particulate concentration (3.29 mg/l) accounting for 82.0% of the total. His value for the average total dissolved hydrocarbon concentration was 400 ug/l, an order of magnitude greater than the values for this study. Different methodology was followed, however, and could be the reason for the variance. The trends for the particulates and dissolved hydrocarbon concentrations, were very similar to ours. The total particulate hydrocarbons concentrations peaked dramatically at the initial surge of the runoff, diminished, and peaked a second time, later in the storm event. The total dissolved hydrocarbon concentrations were initially low and peaked just before or just at the secondary peaking of the total particulate hydrocarbons. The particulate fraction of the water samples possessed the greatest hydrocarbon concentrations. Various studies have shown (Meyers and Quinn, 1973, Khan and Schnitzer, 1972, Meyers and Oas,1978) that particulate material such as clay particles or organic detritus can hold organic species by adsorption and absorption and act as vectors in the movement of these organic species through the hydrologic cycle. This could be one factor in the higher concentrations of hydrocarbons existing in particulate form. In addition, continuous weathering of biogenic and anthropogenic hydrocarbons in the environment result in these hydrocarbons becoming considerably more insoluble in water. As the particulate hydrocarbons are flushed from the discharge channel into the lake, these particulates rain through the water column and deposit themselves onto the sediment layer. Since these forms have been heavily weathered, they could be persistent and incorporate into the sediment column. (Wakeham, 1977; Sheldon and Hites, 1978). Further studies are progressing to model the petroleum hydrocarbon budget of Lake Jackson.

REFERENCES CITED

Blumer, M. and J. Sass. 1972. Indigenous and petroleum-derived hydrocarbons in a polluted sediment. Mar. Pollut. Bull. 3: 92-94.

Giger, W., M. Reinhardt, C. Schaffner, and W. Stumm. 1974. Petroleum-derived and indigenous hydrocarbons in recent sediments of Lake Zug, Switzerland. Environ. Sci. & Tech. 8: 454-455.

Harriss, R.C. and R.R. Turner. 1974. State of Florida. Game and Freshwater Fish Commission. 1973-1974. Job Completion Report. Lake Jackson Investigation. 231 p.

Hunter, J.V., T. Sabatino, R. Gomperts, and M.J. MacKenzie. 1979. Contribution of urban runoff to hydrocarbon pollution. Journal Water Pollut. Control Fed. 51(8): 2129-2138.

Khan, S.U. and M. Schnitzer. 1972. The retention of hydrophobic organic compounds by humic acids. Geochim. et Cosmochim. Acta. 36: 745-754.

Meyers, P.A. and T.G. Oas. 1978. Comparison of association of different hydrocarbons with clay particles in simulated seawater. Environ. Sci. & Tech. 12: 934-937.

Meyers, P.A. and J.G. Quinn. 1973. Factors affecting the association of fatty acids with mineral particles in seawater. Geochim. et Cosmochim. Acta. 37: 1745-1759.

Schamel, S, G.A. Davidson, and T.J. Casey. 1974. "Urban" sediment in Lake Jackson, Florida - a case study in sediment pollution (abstract). Proc. Geol. Soc. Amer. Southeast Section Meeting. April 4, 1974. Atlanta, Ga. p. 396.

Sheldon, L.S. and R.A. Hites 1978. Organic compounds in Delaware River. Environ. Sci. & Tech. 12: 1188-1194.

Smith, S.L. 1972. Lake Management and Research - Lake Jackson Studies. Annual Progress Report, Dingell-Johnson Project F-12-13. Florida Game and Freshwater Fish Commission. Tallahassee. 68 p.

-, 1973. Lake Management and Research - Lake Jackson Studies. Annual Progress Report, Dingell-Johnson Project F-12-13. Florida Game and Freshwater Fish Commission. Tallahassee. 68 p.

- . 1974. Lake Management and Research - Lake Jackson Studies. Annual Progress Report, Dingell-Johnson Project F-12-15. Florida Game and Freshwater Fish Commission. 94 pp.

Turner, R.R., R.C. Harriss, T.M. Burton, and E.A. Laws. 1974. The effect of urben land use of nutrient and suspended solids export from north Florida watersheds. In: Mineral Cycling in Southeastern Ecosystems. Proc. Mineral Cycling Symposium. May 1-3, 1974. Augusta, Ga.

Van Vleet, E.S. and J.G. Quinn. 1977. Input and fate of petroleum hydrocarbons entering the Providence River and Upper Naragansett Bay from wastewater effluents. Environ. Sci. & Tech. 11: 1086-1091.

Wakeham, S.G. 1977. A characterization of the sources of petroleum hydrocarbons in Lake Washington. Journal Water Pollut. Control Fed. 49: 1680-1687.

A COMPARISON OF RAIN-RELATED PHOSPHORUS AND NITROGEN LOADING FROM URBAN, WETLAND, AND AGRICULTURAL SOURCES

R.P. Glandon, F.C. Payne, C.D. McNabb, and T.R. Batterson

Department of Fisheries and Wildlife

Michigan State University

East Lansing, Michigan 48824

ABSTRACT

Comprehensive watershed studies have been conducted for two lakes located in the Lake Michigan drainage system. Studies were conducted from March through October of 1979. During that interval, large differences in storm-related nutrient loading were measured from urban, wetland, and agricultural sources. Eliminating runoff due to melt of the snow pack, it was found that rain-related discharge from the urban area studied was 0.578 kg total-P and 3.688 kg total-N ha^{-1}. Rain induced runoff from marshes in the same drainage basin transported 0.023 kg total-P and 0.585 kg total-N ha^{-1}. Rainfall of approximately the same amount caused runoff from agricultural land of 0.180 kg total-P and 5.965 kg total-N ha^{-1}. Algae of both lakes were phosphorus limited; nitrogen was present in excess. Using constants from Nichols-Dillon relationships in the literature regarding phosphorus, phytoplankton biomass, and secchi disc transparencies, the urban input of phosphorus ha^{-1} of drainage was sufficient to bring 0.96 ha-m of lake water to undesirable algal bloom status. Similarly, marsh input ha^{-1} would bring an estimated 0.04 ha-m into bloom. By the same calculation, storm-related agricultural runoff would result in 0.30 ha-m of lake water becoming undesirably rich in algae. Knowing the number of hectares in these types of catchment and the volume available in a particular lake for phytoplankton production, decisions regarding cost-effective treatment of rain-related discharge can be made.

INTRODUCTION

For a majority of lakes, the most important nutrient factors causing an increase in phytoplankton populations are phosphorus and nitrogen (Vollenweider, 1968). Consequently, a significant portion of the limnological literature has been devoted to the comparison of the annual export of phosphorus and nitrogen from different watershed types (Danigian and Crawford, 1977; Dillon and Kirchner, 1975; Colston, 1974). In general, the studies show that almost every constructional or agricultural activity on a watershed increases nutrient loading in drainage water. The urbanized watershed has been characterized as an outstanding exporter of nutrients (1.10 to 16.60 kg TP $ha^{-1}yr^{-1}$) to surface waters. While untreated sewage water contains the highest nutrient levels, urban storm water runoff alone typically exports nutrients at a rate well above that of other watershed types (Weibel, 1969). Urban fertilization of surface water diminishes valued supplies of suitable quality water for domestic and industrial purposes and frustrates attempts to meet the increased demands for recreational waters.

The potential of watershed runoff to maintain phytoplankton productivity in a receiving lake is largely determined by the level of nutrients exported throughout the summer when temperature and light regimes are optimal. A sustained nutrient input can maintain high productivity by replacing particulate phosphorus and nitrogen sedimented throughout the season (Wetzel, 1975). Annual loading estimates from watersheds in the seasonal north temperate region are often dominated by quantities carried in a spring pulse of discharge. For this reason, annual estimates tend to mask the degree of loading to a lake during the period of stratification.

In this paper, we examine phosphorus and nitrogen losses from marshland, urban, and agricultural watersheds with an emphasis on rain induced loading. This component of surface drainage is used as a means of comparing the watershed types with respect to the ability to maintain phytoplankton productivity in a lake throughout the summer.

STUDY AREAS AND METHODS

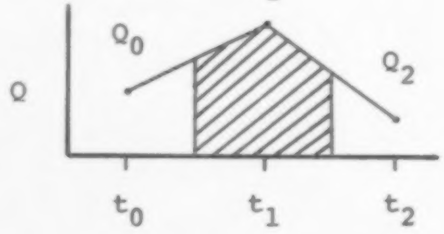
Lake Lansing, Ingham County, Michigan has a low relief watershed of ca.842 hectares. Runoff from approximately 776 ha of the watershed moves to the lake through adjacent marshes and enters the lake through six discrete surface streams. Street drains carry runoff from ca.2.5 ha of residential and commerical areas in the watershed.

Skinner Lake, Noble County, Indiana receives runoff from a rolling agricultural watershed of 3,813 ha. Runoff characteristics were determined from data collected from the Rimmel stream which carries water to the lake from approximately 3,090 ha of the watershed.

Discharge of each marsh stream to Lake Lansing was measured three times per week from the onset of flow (March 5, 1979) to cessation (June 8, 1979). Discharge rates of four of the streams were determined with standard use of a Pigmy-type Gurley current meter. Rectangular weirs were installed across the remaining two streams and discharge was calculated by applying stage height readings to U.S.G.S. approximation formulae (U.S.D.I., 1967).

Water samples were collected from each stream each time discharge was measured and aliquots were composited according to a discharge-proportional scheme devised to reduce the number of samples to be analyzed from 36 (6 stream samples x 3 collections per week x 2 weeks) to 6 (1 two-week discharge-proportional composite per stream). Central to the compositing scheme was the idea that each sample (eg. collected at time t_1) represented the volume of water discharged during the time interval extending from halfway to the previous sample collection ($t_1-t_0/2$), to halfway to the following sample collection ($t_2-t_1/2$). This method accounts for the possibility of detecting changes in water quality resulting from an observed change in discharge.

If the frequency of measurement is high relative to the rate of fluctuation in flow, we can assume a linear change from one measured discharge value to the next. The volume of water discharged represented by the sample collected at t_1 (shaded area in the illustration),



can be calculated:

$$V = \frac{Q_0+3Q_1}{8}(t_1-t_0) + \frac{3Q_1+Q_2}{8}(t_2-t_1), \qquad (1)$$

where, V = the volume of water discharged, and Q_0 = the discharge rate measured at time 0 (t_0). An aliquot from a water sample representing twice the volume of discharge as that of a second sample would be twice as large. Discharge proportional aliquots were added to a composite for intervals of two weeks, after which composite was analyzed for total phosphorus and total nitrogen. "Running" composites were acidified (2 mls conc. H_2SO_4 per liter) and refrigerated. Six composite samples (one per marsh stream) were submitted to the laboratory biweekly.

During the interval of interest, beginning on March 5, the urban drains of the Lake Lansing watershed discharged only during rain events. Relationships between total urban drain discharge and amount of rainfall were determined by plotting the total volume of discharge against the corresponding amount of rainfall for each of several monitored storms. The total volume of water discharged was calculated from periodic measurements of the discharge rate of each drain throughout the monitored storm. Weirs were installed to facilitate the measurement of discharge rates. Water samples collected during the beginning, middle, and end of each monitored storm were composited according to a discharge-proportional scheme analogous to that described for the marsh stream samples. The total amount of rainfall was read from a rain gage set at Lake Lansing.

Discharge of the Rimmel stream in the Skinner Lake watershed was measured daily from 2/23/79 to 4/15/79 with use of a Gurley current meter. Water samples, collected each time discharge was measured, were composited and stored in the same manner as those collected from the Lake Lansing watershed. After 4/15, discharge was measured at two week intervals and grab samples were returned to the laboratory for immediate analysis. Rainfall data were obtained from an automatic precipitation recorder located near the lake.

Total phosphorus concentrations were determined colorimetrically using the single reagent-ascorbic acid method on unfiltered, digested samples.

Total nitrogen is defined as the sum of the nitrate-nitrite and total Kjeldahl nitrogen. Nitrate-nitrite nitrogen was determined colorimetrically by the cadmium reduction method. Total Kjeldahl nitrogen was determined colorimetrically after digestion and distillation of an unfiltered sample. All values are reported as P and N (A.P.H.A., 1975).

RESULTS

The summed discharge rates of the six marsh streams that empty into Lake Lansing reached a maximum of 257 l sec^{-1} on 3/9/79 and fell to zero by 6/13/79 (cf. Figure 1). Early spring fluctuations in the marsh hydrograph correspond to freezing temperatures on 3/11 and from 3/14 to 3/16, and thawing temperatures on 3/19. Thereafter during the spring, the fluctuations in the discharge rate of the marsh correspond to wet and dry periods. During the interval from 3/5 to 3/16 the runoff coefficient,(i.e. m^3 of observed discharge/m of rainfall x m^2 of catchment area) was 1.73, indicating that snow melt was contributing to the observed discharge. From 4/28 to 5/11 the runoff coefficient was 0.36, and from 5/11 to 6/8 the coefficient fell to 0.09.

The peak discharge of 257 l sec^{-1} resulted from the melting snow pack. Separation of runoff due to snow melt from that due to rain events was done by describing the expected recession from a discharge of 257 l sec^{-1} had there been no spring rain. It was assumed that the expected stream flow recession can be characterized by those segments of the hydrograph that were undisturbed by rainfall (Barnes, 1940; Davis and DeWeist, 1960). By isolating dry weather hydrograph segments (eg. from 5/4 to 5/11 of Figure 1), the undisturbed recession could be described by:

$$Q = (a)\ (1.14)^{-t}. \qquad (2)$$

If the recession curve described by this equation is applied to the peak discharge of 3/9, then:

$$Q = (257)\ (1.14)^{-t}, \qquad (3)$$

where, Q is the discharge in liters sec^{-1}, and t is the number of days beyond 3/9. This analysis allows for the separation of total discharge into the component associated with water collected over winter and that induced by spring rain and enables a more meaningful comparison of rain induced runoff between watersheds.

The occurrence of freezing temperatures in the middle of March caused the discharge rate to fall below that expected from the recession curve. This was followed by a warming trend and a spike in discharge, in spite of a lack of rainfall. The area under the curve, between the observed and expected rates of discharge, from 3/10 to 3/16 was used to estimate the volume of water held back by freezing temperatures. This volume of retained water is considered to be a part of the discharge associated with meltwater (cf. stippled areas of Figure 1).

Total phosphorus and total nitrogen content in composited water samples was used with separated components of marsh discharge to yield estimates of nutrient export from the marsh (Table 1). The intervals of measurement correspond to composit-

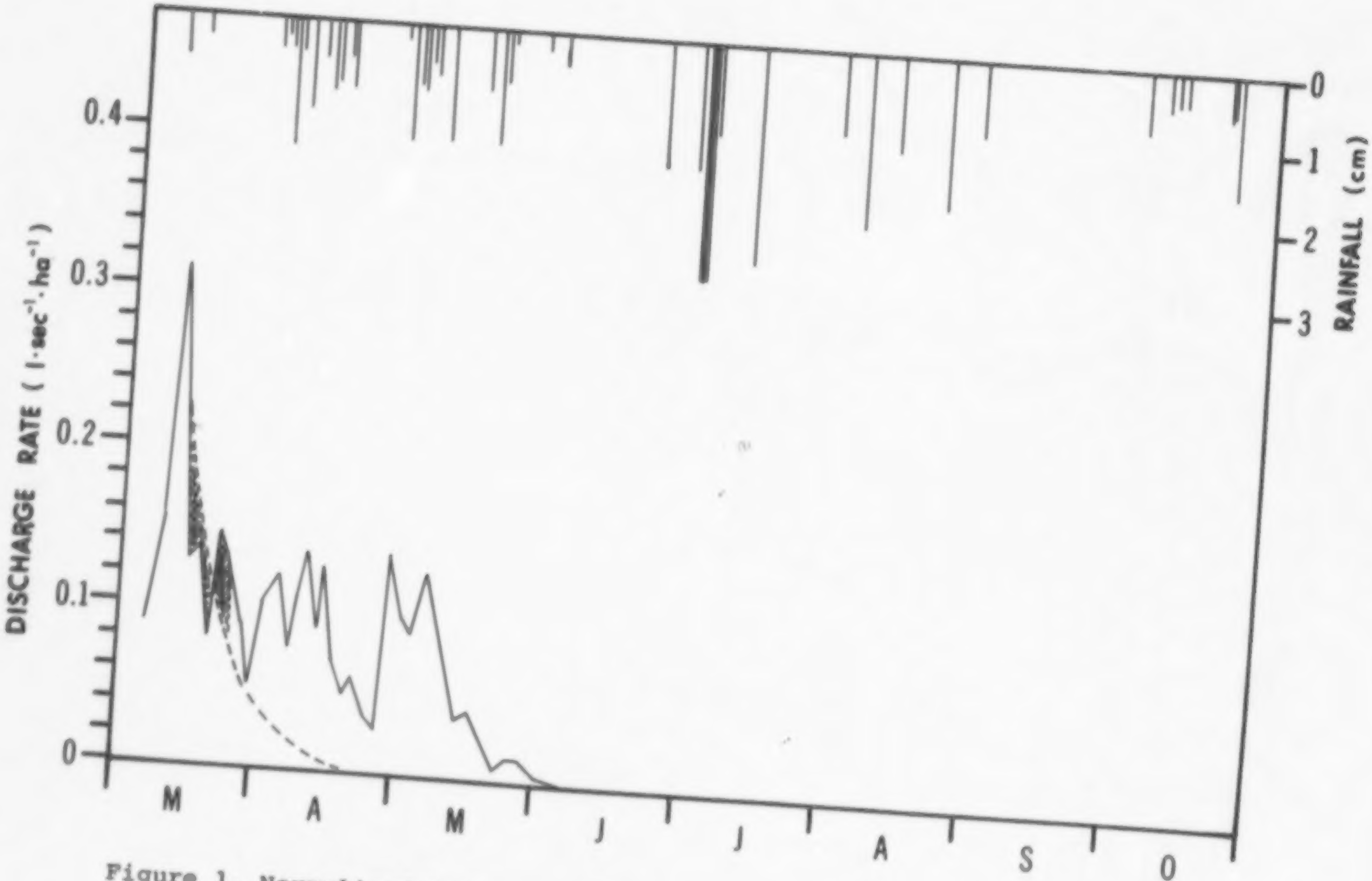


Figure 1. Normalized hydrograph for the summed discharge of six streams draining marshland (ca. 776 ha) in the Lake Lansing watershed and the record of rainfall for the interval March through October of 1979.

Table 1. Total and rain induced loading* of total phosphorus and total nitrogen from marshland in the Lake Lansing watershed (1979).

Interval of Measurement	Total Discharge $m^3 x 10^{-4}$	$m^3 ha^{-1}$	Total Phosphorus kg	kg ha^{-1}	Total Nitrogen kg	kg ha^{-1}
3/5-3/16						
Total	13.4	170	10.5	0.013	166.1	0.214
Rain	nil	-	-	-	-	-
3/17-3/30						
Total	10.8	140	5.7	0.007	134.8	0.174
Rain	3.1	40	1.6	0.002	38.2	0.049
3/31-4/13						
Total	11.6	150	7.9	0.010	141.1	0.182
Rain	9.4	120	6.4	0.008	114.2	0.145
4/14-4/27	7.0	90	4.3	0.006	95.9	0.123
4/28-5/11	9.7	130	4.8	0.006	164.5	0.212
5/12-6/8	2.4	30	0.4	0.001	43.1	0.056
3/5-6/8						
Total	54.9	710	33.6	0.043	745.5	0.961
Rain	31.6	410	17.5	0.023	455.9	0.585

* Rain induced loading = total loading after 4/13.

ing intervals. Estimates of loading of TP and TN were divided by the number of hectares of watershed to normalize the data for comparison. During the entire period of flow, the marshes released a total of 0.043 kg TP ha^{-1} and 0.961 kg TN ha^{-1}. Of that, 0.023 kg TP ha^{-1} and 0.589 kg TN ha^{-1} was released as a result of rainfall during the period.

A comparison of the total discharge from the urban drains during monitored events and the corresponding amount of rainfall reveals two distinct relationships, one associated with the wet spring season and another with summer and fall (Figure 2).

The correlation coefficient (r^2) between urban drain discharge and spring rainfall was 0.87, and the relationship can be described by:

$$V = 6.89 + 7.76x, \tag{4}$$

where, V is the total volume of discharge from a spring storm and x is the corresponding amount of rainfall in mm.

The discharge-rainfall relationship in the summer and fall reflects a lower runoff coefficient. This relationship exhibited a correlation coefficient of 0.96, and is described by:

$$V = -6.45 + 4.73x. \tag{5}$$

The linear description of the spring relationship was applied to the local rainfall record from 3/5 to 6/8 to estimate the volume of urban drain discharge. The summer-fall relationship was applied to the amount and occurrence of rainfall in the interval from 6/9 to 10/22 (Table 2).

Total phosphorus and total nitrogen content in composite samples collected during monitored events averaged 0.547 mg TP l-l and 3.493 mg TN l^{-1} with one standard deviation of 0.149 and 1.120 respectively. There was no statistically significant difference (α<0.05) between the elemental concentrations in spring and summer-fall samples. The average TP and TN levels of all samples were used with urban drain discharge to calculate nutrient export.

For the entire season under consideration (3/5 to 10/22), the per ha loading of TP was 25 times the rain induced marsh loading. Urban drains carried nearly 6 times the rain induced TN loading from the marsh.

The discharge hydrograph of the Rimmel stream in the Skinner Lake watershed shows a dramatic elevation in discharge rate during the first week in March and falls to a stable flow by the middle of May (Figure 3). Separation of the Rimmel hydrograph was done in a manner similar to that of the marsh hydrograph. Dry weather segments (eg. 4/12 to 4/15) were con-

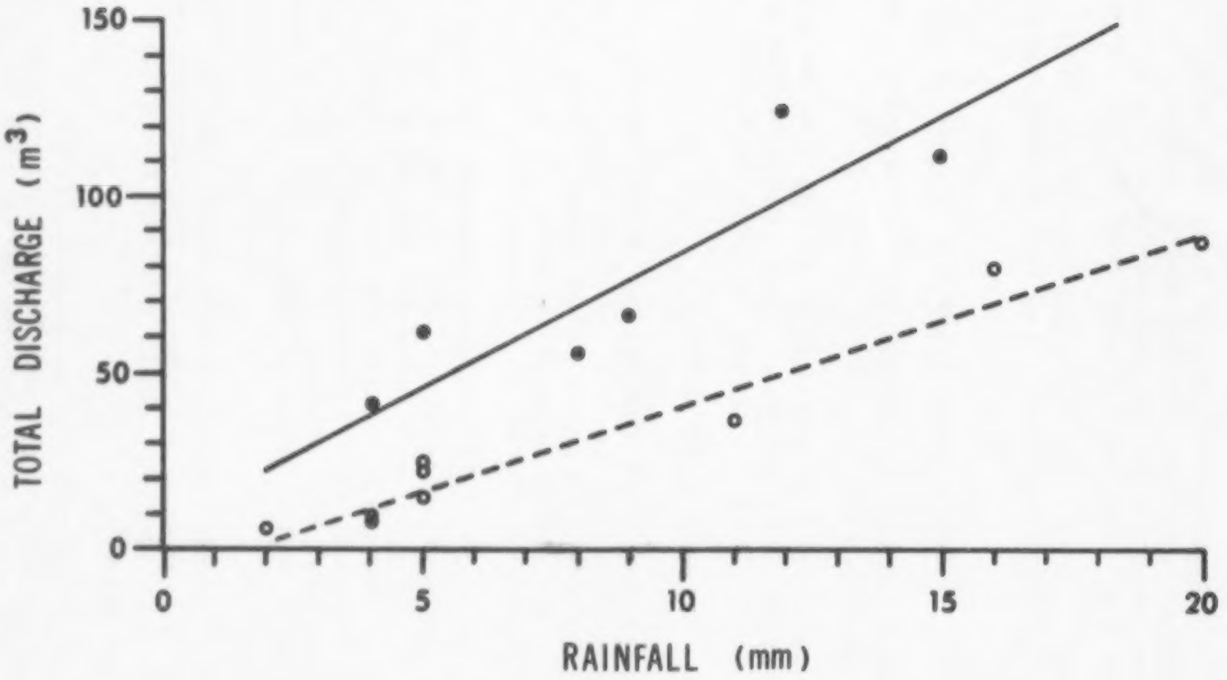


Figure 2. Relationship between rainfall and summed discharge from urban (street) drains emptying into Lake Lansing in the spring (——) and summer-fall (----) periods of 1979.

Table 2. Rain induced loading of total phosphorus and total nitrogen to Lake Lansing from urban street drains (1979).

Interval of Measurement	Total Discharge		Total Phosphorus		Total Nitrogen	
	m^3	$m^3 ha^{-1}$	kg	kg ha^{-1}	kg	kg ha^{-1}
3/5-3/16	56	22	0.031	0.012	0.196	0.078
3/17-3/30	177	71	0.097	0.039	0.618	0.247
3/31-4/13	343	137	0.188	0.075	1.198	0.479
4/14-4/27	225	90	0.123	0.049	0.786	0.314
4/28-5/11	311	124	0.170	0.068	1.086	0.434
5/12-6/8	399	160	0.218	0.087	1.394	0.558
6/9-6/23	139	56	0.076	0.030	0.486	0.194
6/24-7/8	437	175	0.239	0.096	1.526	0.610
7/9-7/23	132	53	0.072	0.029	0.461	0.184
7/24-8/8	142	57	0.078	0.031	0.496	0.198
8/9-8/23	143	57	0.078	0.031	0.499	0.200
8/24-9/7	42	17	0.023	0.009	0.147	0.059
9/8-9/29	0	0	-	-	-	-
9/30-10/14	76	30	0.042	0.017	0.265	0.106
10/15-10/22	18	7	0.010	0.004	0.063	0.025
3/5-6/8	1511	604	0.827	0.331	5.278	2.111
3/5-10/22	2640	1056	1.445	0.578	9.221	3.688

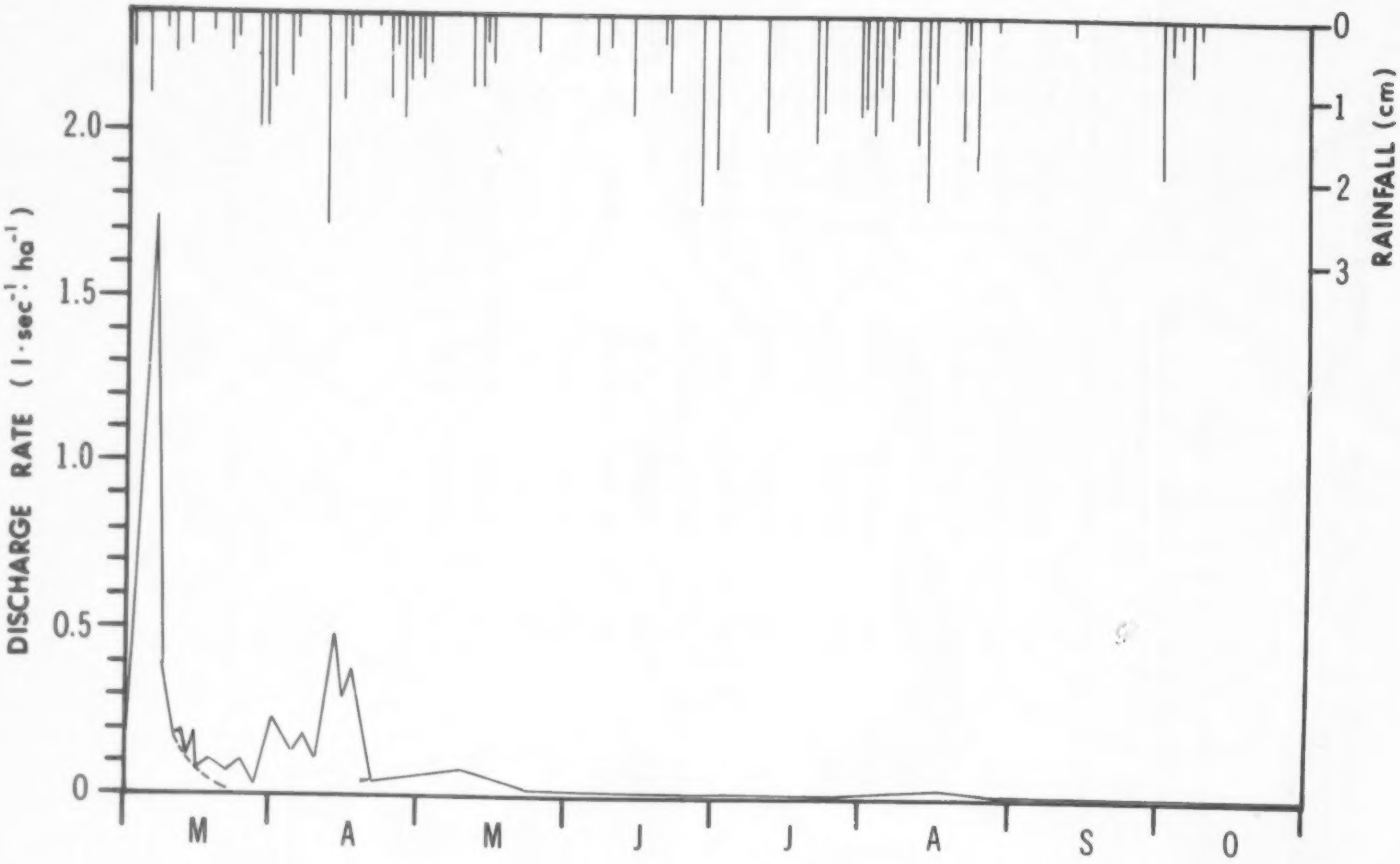


Figure 3. Normalized hydrograph for the Rimmel stream draining agricultural land (ca. 3090 ha) in the Skinner Lake watershed and the record of rainfall for the interval March through October of 1979.

sidered in order to describe the expected rate of recession of discharge in the absence of rainfall. The expected decay of the discharge rate from the level observed on 3/6 can be described by:

$$Q = (1.25)\ (0.74)^{t}, \tag{6}$$

where, t is the number of days after 3/6.

TP and TN concentrations in composited samples were used with separated components of discharge to calculate nutrient export from the agricultural land (Table 3). The rain induced loading over the 2/23 to 10/22 interval was 0.180 kg TP ha^{-1} and 5.965 kg TN ha-1.

In summary, the normalized rain induced loading from 2/23/79 to 10/22/79 in kg ha^{-1} was:

$$\text{TP:}\quad 0.023_{\text{(marsh)}} < 0.180_{\text{(agriculture)}} < 0.587_{\text{(urban)}}$$

$$\text{TN:}\quad 0.585_{\text{(marsh)}} < 3.688_{\text{(urban)}} < 5.965_{\text{(agriculture)}}.$$

DISCUSSION

In comparison, the marshland discharge had the lowest capacity to support phytoplankton productivity throughout the summer based on the low level of total nutrient contribution and the early-summer termination of discharge. By contrast, the urban and agricultural watersheds exported roughly 10 times the nitrogen and phosphorus in sustained loading throughout the summer and fall. The significance of this nutrient loading can be given perspective by examining the potential impact on populations of phytoplankton in a receiving lake.

Other studies of nutrient levels in surface waters have shown that phosphorus concentrations are generally the phytoplankton yield-limiting factor (Edmondson, 1972; Schindler, 1974). As a result, a number of empirically defined relationships between measured phosphorus and the abundance of phytoplankton in lakes have been reported (Dillon and Rigler, 1974). They are intended for use as predictive tools. Through an examination of the available literature, Nichols and Dillon (1978) have found that the most well defined relationship is that between summer total phosphorus concentration in the euphotic zone of a lake and the average summer phytoplankton cell volume. The relationship between these variables is linear and can be described by:

$$\text{phytoplankton cell volume} = 0.21x - 2.6, \tag{7}$$

where, x is the total phosphorus concentration in mg m^{-3}, and

Table 3. Total and rain induced loading* of total phosphorus and total nitrogen from agricultural land in the Skinner Lake watershed (1979).

Interval of Measurement	Total Discharge		Total Phosphorus		Total Nitrogen	
	$m^3 x 10^{-5}$	$m^3 ha^{-1}$	kg	$kg\ ha^{-1}$	kg	$kg\ ha^{-1}$
2/23-3/6						
Total	11.2	362	478	0.155	8362	2.705
Rain	4.6	150	198	0.064	3468	1.122
3/7-3/31						
Total	11.1	358	285	0.092	9313	3.013
Rain	3.5	114	90	0.029	2957	0.957
4/1-4/15	9.6	311	170	0.055	7613	2.463
4/16-4/30	3.2	103	21	0.007	1295	0.419
5/1-5/14	2.6	84	20	0.006	1291	0.418
5/15-5/28	0.9	30	8	0.002	310	0.100
5/29-6/12	0.5	15	2	0.001	136	0.044
6/13-6/25	0.4	14	1	0.000	179	0.058
6/26-7/9	0.4	14	3	0.001	245	0.079
7/10-7/23	0.4	13	4	0.001	110	0.035
7/24-8/5	0.5	15	1	0.000	81	0.026
8/6-8/20	0.8	25	14	0.005	348	0.113
8/31-9/6	0.8	27	8	0.003	96	0.031
9/7-9/25	0.9	29	6	0.002	177	0.057
9/26-10/13	0.9	28	10	0.003	128	0.041
2/23-10/13						
Total	44.2	1428	1031	0.333	29556	9.602
Rain	30.0	972	556	0.180	18306	5.965

* Rain induced loading = total loading after 3/31.

the cell volume is expressed in $mm^3\ l^{-1}$. Nichols and Dillon further present a well defined relationship between cell volume and resultant secchi disc visibility. This relationship was shown to be non-linear, and can be generally described by:

$$\text{secchi disc visibility} = (3.20)\ (1.08)^{-x},$$

where, x is the phytoplankton cell volume and secchi disc visibility is expressed in meters.

Using these relationships and selecting a measure of nutrient-related water quality that is undesirable, for example a secchi disc transparency of 1.5m or less, the impact of nutrients discharged to lakes from urban, marsh, or agricultural land can be predicted. The Nichols-Dillon relationships are such that at a secchi disc measurement of 1.5m, the predicted volume of phytoplankton cells would be near $10mm^3\ l^{-1}$. This corresponds to a total phosphorus concentration of 60 mg m^{-3}.

If this level of phytoplankton density is chosen as an index of minimum acceptable water quality, then by our data the rain induced total phosphorus loading from one hectare of marshland would be sufficient to degrade 0.04 ha-m of phosphorus limited lake volume to that level. By the same calculation, the discharge from one hectare of the agricultural watershed would adversely affect 0.30 ha-m of lake volume. By contrast, summer runoff from the urban watershed would debase 0.96 ha-m of lake volume, roughly 3 times the impact of the agricultural watershed and 24 times that of the marshland.

Assuming that watershed export of phosphorus increases linearly with the size of the watershed, comparisons like these can be used to predict the impact on a lake that would occur if a watershed were converted from one type to another. For example, our export estimates from marsh, agricultural, and urban areas were used with the Nichols-Dillon relationships and the criterion that a secchi disc transparency of 1.5m or less constitutes undesirable nutrient-related water quality, to predict the volume of phosphorus limited lake water adversely impacted by urbanizing either marsh or agricultural land. The result is Figure 4.

The figure shows that for a given watershed area, the volume of receiving water adversely impacted (minimum receiving volume) increases to an upper limit as urbanization is increased towards 100%. In approaching the upper limit, the figure suggests that a given increase in urbanization of marshland would cause a greater change over previous conditions than would the same increase in urbanization of agricultural land.

While our data are site specific to a degree, the technique has general applicability. The approach can be used as a basis for decisions regarding cost effective treatment of existing rain-related discharges, or decisions regarding the need for

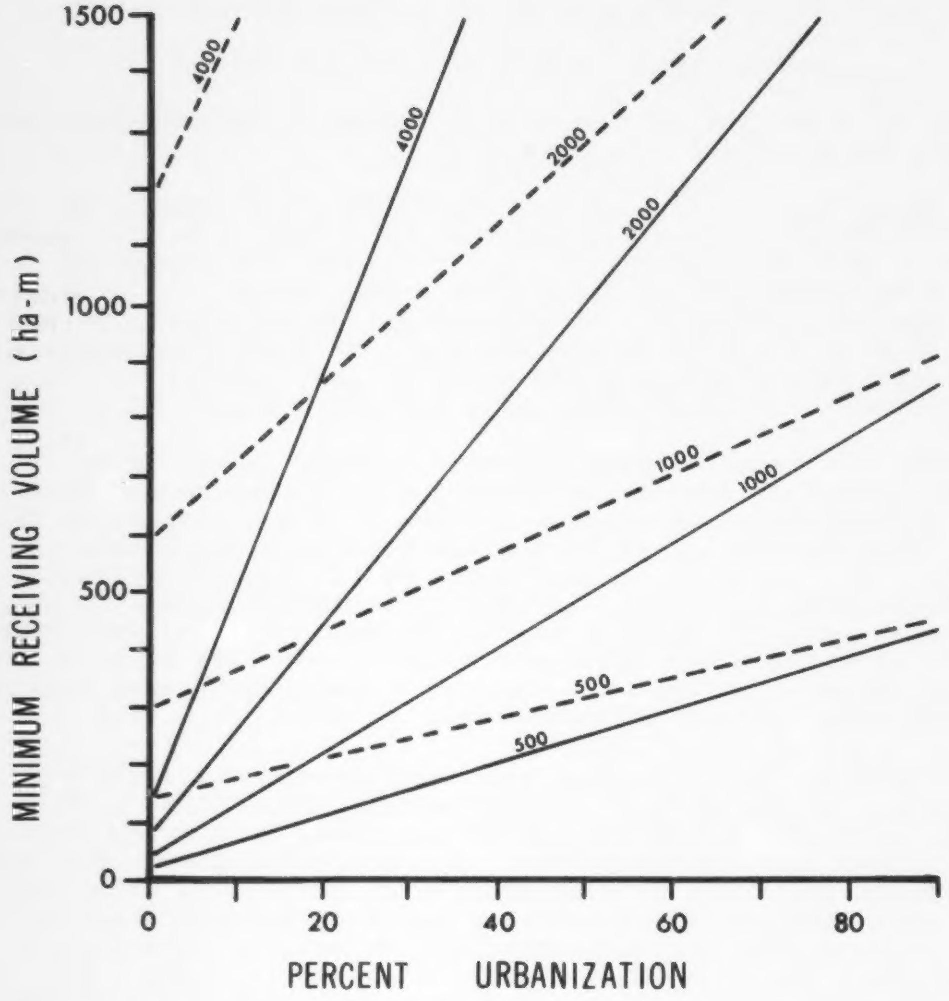


Figure 4. The minimum volume of phosphorus limited lake water that would be brought to an undesirable phytoplankton density by urbanizing marshland (——) or agricultural land (----) according to the data of this study. Numbers on lines indicate selected marsh and agricultural drainage areas in hectares.

treating discharges from urban developments during the planning stages of those developments.

ACKNOWLEDGEMENTS

This work has been supported by the U.S. Environmental Protection Agency, Office of Water Planning and Standards, as Grant No. R80504601 administered by the Corvallis Environmental Research Laboratory as a part of the evaluation phase of the Clean Lakes Program, and by the Michigan Agricultural Experiment Station. John R. Craig, George W. Knoecklein, John M. McCabe, Maureen M. Wilson, G. Douglas Pullman and Mehdi Siami of the Limnological Research Laboratory at Michigan State University provided valuable assistance.

LITERATURE CITED

American Public Health Association. 1975. Standard Methods for Examination of Water and Wastewater. 14th ed. American Public Health Assoc., Washington, D.C. 1193 pp.

Barnes, B.S. 1940. Discussion of analysis of runoff characteristics by O.H. Meyer. Amer. Soc. Civil Eng. Trans. 105:1-106.

Colston, N.V. 1974. Characterization and treatment of urban land runoff. EPA-670/2-74-096. U.S. Environmental Protection Agency, Cincinnati, Ohio. 170 pp.

Danigian, A.S., and Crawford, N.H. 1977. Simulation of nutrient loading in surface runoff with the NPS model. EPA-600/3-77-065. U.S. Environmental Protection Agency, Office of Res. and Dev., Athens, Georgia. 110 pp.

Davis, S.N., and DeWiest, R.J. 1966. Hydrogeology. John Wiley and Sons, Inc., New York. 463 pp.

Dillon, P.J., and Rigler, F.H. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 17:250-254.

Dillon, P.J., and Kirchner, W.B. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Res. 9:135-148.

Edmondson, W.T. 1972. Nutrients and phytoplankton in Lake Washington. In G.E. Likens, ed. Nutrients and Eutrophication: The Limiting-Nutrient Controversy. Special Symposium, Amer. Soc. Limnol. Oceanogr. 1:172-193.

Nichols, K.H., and Dillon, P.J. 1978. An evaluation of phosphorus-chlorophyll-phytoplankton relationships for lakes. Int. Revue ges. Hydrobiol. 63:141-154.

Schindler, D.W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. Science, 184:897-899.

U.S. Department of the Interior. 1967. Water Measurement Manual. 2nd ed. U.S.D.I., Washington, D.C. 329 pp.

Vollenweider, R.A. 1968. Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication. Rep. Organisation for Economic Cooperation and Development, DAS/CSI/68.27, Paris. 192 pp.; Annex, 21 pp.; Bibliography, 61 pp.

Weibel, S.R. 1969. Urban drainage as a factor in eutrophication. In Eutrophication: Causes, Consequences, Correctives. National Academy of Sciences, Washington, D.C., pp. 383-403.

Wetzel, R.G. 1975. Limnology. W.B. Saunders Co., Philadelphia. 743 pp.

4

Third Session

IMPACTS ON LAKES AND RIVERS

Moderator: Francis J. Condon
EPA, Washington, D.C.

IMPACTS OF STORMWATER RUNOFF ON A FLORIDA LAKE ECOSYSTEM: EFFECTS ON WATER QUALITY AND BIOTA

by

Eldon C. Blancher II
Marine Environmental Sciences Consortium
P. O. Box 386
Dauphin Island, Alabama 36528

ABSTRACT

A study of external nutrient loadings to the Lake Conway ecosystem, an interconnected series of three lakes located in Orange County, Florida, showed that both nitrogen (2.6 g-N/m^2-yr) and phosphorus (0.22 g-P/m^2-yr) inputs were within the range of loadings that leads to mesotrophic conditions. The major external sources of both elements were atmospheric inputs, stormwater runoff and subsurface seepage.

Multivariate analysis of water quality data by discriminant analysis showed differences among the three lakes of the Conway system. Seasonal trends in several water quality indicators varied concomitantly with changes in external nutrient loadings, especially with those from residential stormwater runoff. Those lakes that had a proportionately larger share of phosphorus loadings from stormwater runoff showed a degradation in water quality. Strong linear relationships were found between watershed area to lake area ration and stormwater phosphorus loadings ($r^2 = 0.98$); stormwater phosphorus loadings and Secchi disk transparency ($r^2 - 0.88$) and chlorophyll a and total zooplankton numbers ($r^2 = 0.88$). These findings indicate a direct impact of stormwater runoff on water quality and subsequently on changes in the lakes biota.

INTRODUCTION

Numerous receiving water bodies have been adversely affected by the introduction of wastewaters from various sources. While many investigations have shown the direct impact of point source pollution, (for instance, Edmondsons' (1972) study as the recovery of Lake Washington after sewage diversion) relatively few have shown the direct effect on non-point source pollution, particularly stormwater, on water quality and biota. A large number of excellent studies have recently been made on the quantity and quality of stormwater runoff but in the majority of these studies the actual impact of this pollution source can only be inferred. Both the paucity of comparative data to systems receiving varying pollutional loadings and the confounding influence of additional pollution sources have contributed largely to this lack of conclusive studies.

Ideally, to test the hypothesis of the impact of stormwater runoff, one should identify a group of receiving water bodies with similar physical, chemical and biological features which vary in stormwater loadings and are relatively free of other pollution inputs. Rarely do reasearchers have the opportunity to work on such "ideal" systems. Fortunately, the Lake Conway system discussed in this paper did meet the above criteria and served as an excellent test case for demonstrating the impacts of stormwater runoff to a Florida Lake ecosystem.

DESCRIPTION OF THE STUDY AREA

The Lake Conway system is comprised of a series of fused dolines (Figure 1). Average depth for the system is 5.2 meters and surface areas range from 0.28 km^2 for Lake Gatlin to almost 3.0 km^2 for the middle pool (Table 1). These lakes are fairly representative of typical solution basin lake found in Florida. The surrounding areas in the watershed (Table 1) can be classified as primarily residential, among which are interspersed areas classified as agricultural, mainly citrus. However, few of these citrus areas have significant agricultural activities and they could probably be better classified as undeveloped land. A number of studies recently have been published on the Lake Conway system in conjunction with the U. S. Army's Corps of Engineers Project. These studies include a general description of the Lake Conway area (Theriot 1977), and studies of its fishes (Guillory _et al_. 1977), aquatic macrophytes (Nall _et al_. 1977, 1978), plankton and benthos (Fox _et al_. 1977; Conley _et al_.), water quality (Sawicki 1977), as well as system modeling (Ewell and Fontaine 1977, Fontaine and Ewell 1978), nutrient budgets (Blancher _et al_. 1978; Blancher and Fellows 1979), and a study of the nitrogen cycle (Sompongse 1978). The reader is referred to these articles for an additional discussion of these topics.

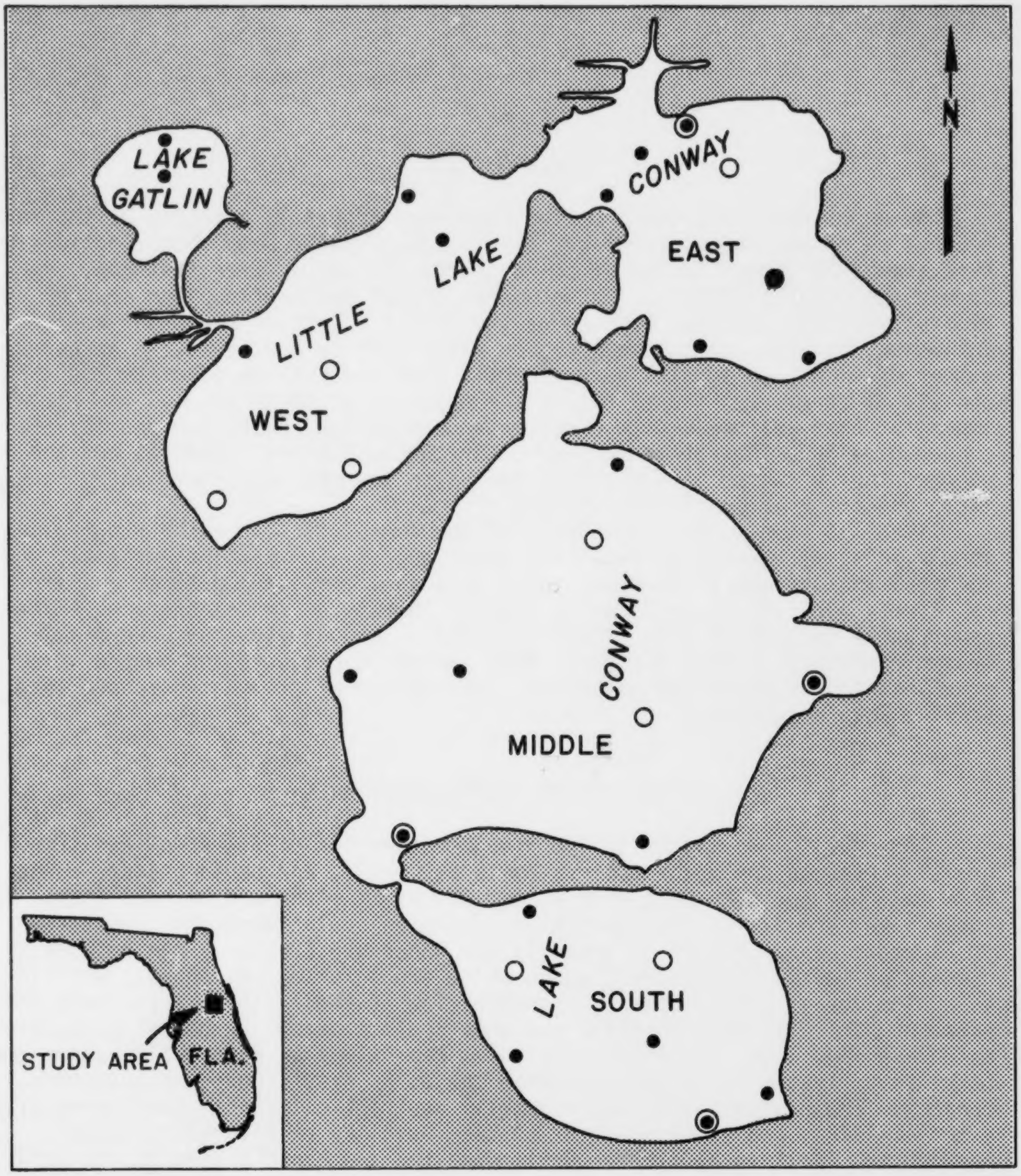


Figure 1. A map of the Lake Conway system indicating the location of sampling stations. (0) indicates stations sampled from April 1976 through March 1978, (0) those sampled from April 1976 through March 1977, and (0) those sampled from April 1977 through March 1978.

Table 1. FEATURES OF THE LAKE CONWAY SYSTEM AND ASSOCIATED DRAINAGE BASIN[a].

	Area (km^2)	Mean depth (meters)	Total Watershed Area	Watershed Area (less lake)	% Residential	% Citrus or Undeveloped Areas[b]
South pool	1.37	6.01	3.12	1.75	77.3	22.6
Middle pool	2.99	5.98	6.14	3.15	68.2	31.8
East pool	1.27	4.54	11.19	9.92	50.9	43.40
West pool	1.48	5.72	10.94	9.46	69.3	14.3
Gatlin	.28	4.18	3.41	3.13	74.8	11.1
Total	7.39	5.29	34.80	27.41	68.1	29.2

[a] Areas reported in km^2.

[b] Balance of the watershed area includes ponds and other depression storage areas.

METHODS AND MATERIALS

HYDROLOGY

Lake and watershed areas were calculated from United States Geological Survey topographical maps with an electronic planimeter (Hewlett Packard 9810A-9864A). Extent of catchment area was determined from topographic features and by consulting area tax assessment maps at the Orange County Public Works Office for street drainage patterns. Land use and development patterns for the watershed were estimated from a recent (1975) aerial photograph supplied by county engineers (Orange County Public Works, Orlando, FL) and updated by visual reconnaisance.

Monthly rainfall and evaporation data were obtained from U. S. Weather Bureau climatological data reports (NOAA 1972-1978). Rainfall data reported by the Orlando weather station at McCoy Air Base were used because of the close proximity of the airport to the lake.

NUTRIENT INPUTS

Aolean sources were monitored utilizing a wet/dry precipitation collector placed on a boat house extending over the lake. Basically, the unit consisted of two large buckets with a servo-operated lid and a precipitation sensor.

Since there were no point sources entering the lake, our efforts were directed to determining the non-point sources such as stormwater runoff from the residential areas. The rational method was used, calibrated by monitoring the quantity and quality of runoff using a v-notch weir in conjunction with an automated flowmeter and sampler unit. The particular unit used had a bubbler type flow meter which activated the sampler based on flow rate. Thus flow weighted averages could be computed. Subsurfaces seepage measurements were made by Fellows (1978) utilizing the methods of Lee (1977). Top portions of 55 gallon drums were pushed into the sediments and allowed to equilibrate for several weeks. Water was collected in collapsed latex bags to determine seepage flow and then analyzed for nutrients.

WATER QUALITY AND BIOTA

Secchi disk transparency, dissolved oxygen, temperature, pH, chlorophyll a, conductivity, turbidity, total hardness, and total alkalinity were routinely measured at all station in the Lake Conway system. Monthly primary productivity measurements by the light-dark bottle method (APHA, 1975) were made at 1 meter depth in the center of each pool of the Lake Conway system from June 1976 through March 1978. Nutrient samples were obtained from each of the pools from June 1976 through March 1978 in conjunction with biologic sampling. Additional nutrient information was supplied by the Orange County Pollution Control department. All analyses were perforned using standard methods (APHA, 1975; U. S. EPA 1976) and average nutrient concentrations were computed by weighting the samples on an areal basis.

Analysis of the trophic indicators of the Conway system were accomplished using multivariate techniques. The indicators used in this study were conductivity, total phosphorus, total nitrogen, Secchi disk transparency, functional chlorophyll a, and primary productivity. Ordination of the various pools according to trophic indicators was accomplished using stepwise discriminate analysis (BMDP7M).

Monthly zooplankton samples were collected at littoral (< 3 m) and limnetic (> 3 m) stations in each of the five pools of the Lake Conway system following two sampling regimes. From April 1976 to March 1977, 16 littoral and 5 limnetic stations were sampled, and from April 1977 to March 1978, 13 littoral and limnetic stations were utilized (Figure 1).

Zooplankton samples were collected at all stations by a vertical haul with a U. S. Standard # 10 (153 μ mesh) Wisconsin plankton net. Data obtained by this method were expressed on both an areal and a volumetric basis.

RESULTS AND DISCUSSION

HYDROLOGIC AND NUTRIENT LOADINGS

Nutrient inputs to a lacustrine system are closely related to the hydrologic budget of the system. Thus, critical to the development of any realistic materials budget is the construction of an accurate water budget. Every attempt was made to insure that all possible sources of nutrient inputs were assessed. For the sake of brevity, only the most important sources are considered here. Detailed discussion of all possible inputs to the system have been previously presented (Blancher 1979).

Stormwater hydrographs for the Conway basin were calculated from data collected on August 1976 and March, May, and June 1978. Measurements of stormwater flows during 1977 were not possible due to the paucity of rainfall. Of the precipitation that fell during those storms, 3.5 percent on an average entered the lake as stormwater runoff. Initiation of runoff and time interval to peak flow were generally rapid as was return to base flow.

In situ seepage measurements provide direct measurements of seepage flow. Prior to the development of this technique, seepage flow was estimated by subtracting the measurable outputs from the inputs. The seepage meter method used by Fellows (1978) provided values for seepage with an estimated error margin of ± 20 percent.

Hydrologic inputs for the individual pools of the Lake Conway system are presented in Table 2. Precipitation is the dominant hydrologic input of all the pools. Seepage for the various pools varies from 15 to 23 percent whereas inputs from stormwater varies from 3 percent in the south pool to 28 percent in Lake Gatlin.

In order to verify the accuracy of the parameters used in the hydrologic budget, the calculated flows were used in a dynamic model to predict change in storage within the total lake system. An information flow diagram for the model is presented in Figure 2. The model was simulated for the period from December 1972 through March 1978 on a monthly basis.

The results of the model are compared with the actual changes in lake height in Figure 3.

Differences between the simulated and observed changes in lake height are within five percent on a volume basis. For a detailed discussion of the development of the model see Blancher (1979).

Aolean nutrient loadings based on the rainfall and dry fallout samples

Table 2. HYDROLOGIC INPUTS FOR THE INDIVIDUAL POOLS IN THE CONWAY SYSTEM*. NUMBERS IN PARENTHESES INDICATE THE PERCENT OF TOTAL INFLOW IN THAT POOL.

Pool	Precipitation	Stormwater	Seepage In	Total In
South	1.64 (78)	.076 (4)	.395 (19)	2.11
Middle	3.59 (82)	.136 (3)	.669 (15)	4.39
East	1.52 (58)	.626 (24)	.488 (19)	2.63
West	1.78 (63)	.629 (22)	.433 (15)	2.84
Gatlin	.34 (49)	.197 (28)	.164 (23)	0.70

*In millions of m^3.

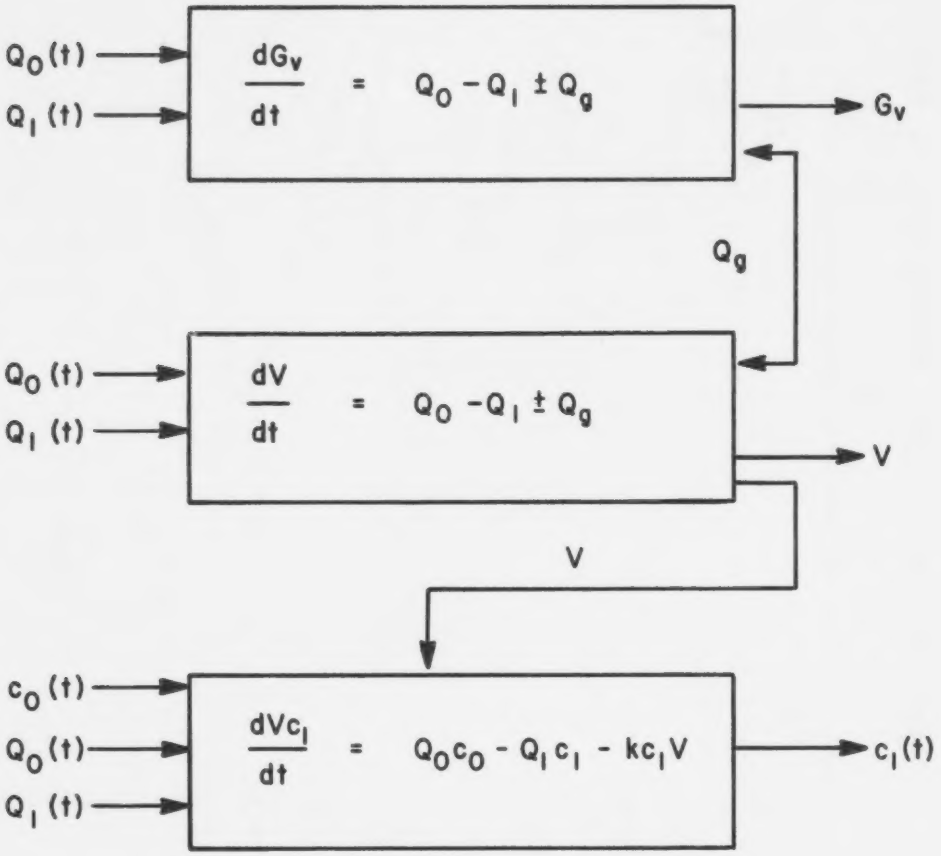

$Q_0(t)$
$Q_1(t)$
$\frac{dG_v}{dt} = Q_0 - Q_1 \pm Q_g$
G_v
Q_g
$Q_0(t)$
$Q_1(t)$
$\frac{dV}{dt} = Q_0 - Q_1 \pm Q_g$
V
V
$c_0(t)$
$Q_0(t)$
$Q_1(t)$
$\frac{dVc_1}{dt} = Q_0c_0 - Q_1c_1 - kc_1V$
$c_1(t)$

Figure 2. An information flow diagram for the dynamic hydrologic-materials systems model.

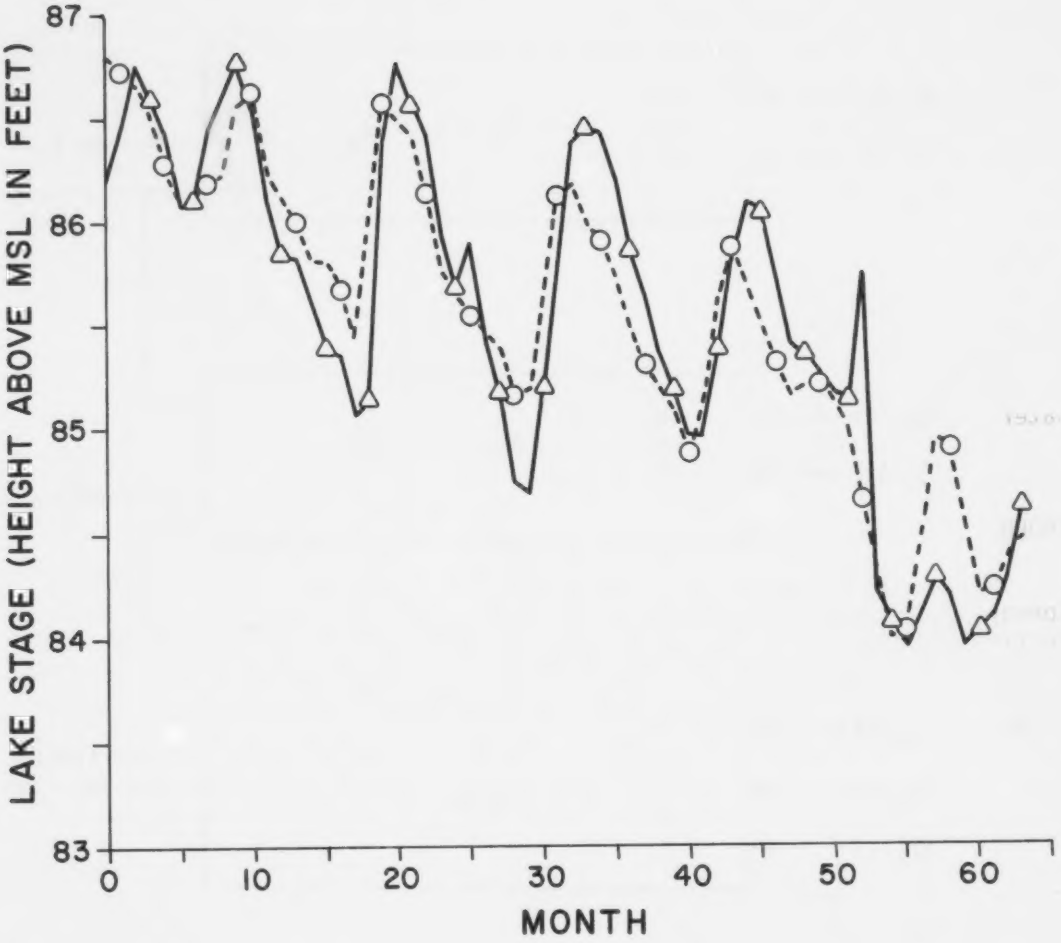


Figure 3. Comparison of the actual variations in lake height of the Lake Conway System (Δ) and the simulated height from the hydrologic model (o) for the period December 1972 (month = 0) through March 1978 (month = 63).

collected, provided annual loading values of 0.048 gP/m^2-yr and 0.36 gN/m^2-yr for rainfall and 0.080 gP/m^2-yr and 0.57 gN/m^2-yr for dry fallout. The resulting bulk loadings (0.125 gP/m^2-yr and 0.93 gN/m^2-yr) compare well with similar loadings of 0.105 gP/m^2-yr and 1.0 gN/m^2-yr for Gainesville Florida (Hendry and Edgerton 1978, unpublished data, University of Florida, Dept. of Environmental Engineering).

Analysis of nitrogen and phosphorus concentrations in stormwater runoff from a small catchment in the Lake Conway watershed yielded average concentrations of 3.25 mg/L and 0.39 mg/L for nitrogen and phosphorus, respectively (Table 3). These compare well to published values which range from 1.93 to 4.45 gm/L for N and from 0.19 to 0.98 mg/L for P (Weibel 1969; Kluesener and Lee 1974; Mattraw and Sherwood 1977).

Nutrient inputs from subsurface seepage to Lake Conway were determined by Fellows (1978). Annual loadings per meter of Lake shoreline from this source, calculated as an average of three sites were found to be 0.66 gN/m-yr and 0.024 gP/m-yr.

Additional external sources of nutrients were identified but accounted on the average for less than 5 percent of the inputs to the lakes and hence were considered insignificant.

Areal nitrogen and phosphorus loading rates from the three major external sources are presented in Table 4. Of these input sources, the stormwater loading rate was most variable between the pools accounting for approximately 10 percent of the N and P inputs in the South pool to about 50 percent in Lake Gatlin.

TROPHIC STATE/WATER QUALITY VARIATIONS

Six parameters were chosen as trophic/water quality indicators of the Conway system representing the physical (Secchi disk transparency and conductivity), chemical (total nitrogen and phosphorus), and biolgoical (functional chlorophyll _a_ and gross primary productivity) attributes of the lakes. Comparisons between those parameters were used as a basis of determining difference in water quality.

Limnologists have determined general guidelines for determining trophic state based on total phosphorus, chlorophyll _a_, and Secchi disk transparency. Gakstatter _et al_. (1975) reviewed the literature and established criteria used for classifying the lakes sampled for the National Eutrophication Survey (NES). Comparing their values with annual average values of the five pools of the Conway system (Table 5) we see that the south and middle pools border on oligotrophy, whereas the east and west pools are decidedly mesotrophic. Using the NES criteria, Lake Gatlin would be considered eutrophic. While this comparison illustrates the overall (average) trophic states of these lakes, better resolution between the study lakes would be more useful in examining the mechanisms and causative factors involved in the eutrophication process. This resolution is necessary for any comparisons of the importance of various loading sources to overall lake response.

The multivariate method of discriminant analysis provides linear functions of the variables from a multidimensional data set. These functions then allow the classification of the multidimensional data vector into one of several multivariate normal populations. A modified procedure known as stepwise discriminant analysis (BMDP7M, Dixon and Brown 1977, using default options) was used to identify the variables that would produce a "best-fit"

Table 3. NITROGEN AND PHOSPHORUS CONCENTRATIONS IN STORMWATER RUNOFF ENTERING LAKE CONWAY. CONCENTRATIONS ARE FLOW WEIGHTED AVERAGES EXPRESSED AS g/m^3.

	Number of Samples	Total Nitrogen	Total Phosphorus	Total Flow (m^3)
21 August 1976	9	2.3	0.33	986.3
23 March 1978	27	4.7	0.45	45.5
4 May 1978	28	4.6	0.47	665.8
16 June 1978	12	4.0	0.42	45.8
21 June 1978	8	5.1	0.63	4.6
Mean Values (flow weighted)	--	3.25	0.39	--

TABLE 4. AREAL NITROGEN AND PHOSPHOROUS LOADING RATES FROM MAJOR EXTERNAL SOURCES. RATES REPORTES IN g/m^2-yr.

	Aolean		Storwater		Seepage		Total		Percent Loading From Stormwater	
	N	P	N	P	N	P	N	P	N	P
South	0.93	0.125	0.13	0.015	0.62	0.024	1.68	0.164	8	9
Middle	0.93	0.125	0.10	0.012	0.48	0.019	1.51	0.156	7	8
East	0.93	0.125	1.11	0.134	0.83	0.032	2.90	0.291	39	46
West	0.93	0.125	0.96	0.116	0.63	0.025	2.52	0.266	38	44
Gatlin	0.93	0.125	1.94	0.232	1.27	0.049	4.14	0.406	47	57

Table 5. NATIONAL EUTROPHICATION SURVEY GUIDELINES FOR DETERMINING TROPHIC STATE (AFTER GAKSTATTER ET AL. 1975) AND MEANS OF SEVERAL TROPHIC STATE INDICATORS FROM THE VARIOUS POOLS OF THE CONWAY SYSTEM.

	Oligotrophic	Mesotrophic	Eutrophic	South Pool	Middle Pool	East Pool	West Pool	Lake Gatlin
Total Phosphorus (mg/L)	< 0.01	0.01-0.02	> 0.02	0.016	0.018	0.021	0.020	0.021
Chlorophyll a (μg/L)	< 4.0	4-10	>10.10	2.8	3.7	4.14	4.47	9.49
Secchi depth (m)	> 3.7	2.0 -3.7	< 2.0	3.46	3.30	2.70	2.66	1.86

model to discriminate between the populations. Each of the pools in the Conway system was treated as a "population", and the observed variables included the monthly mean values of the six trophic state indicators. The procedure was performed using all six indicators, as well as several subsets of the variables.

The results of the analyses (Table 6) show that conductivity and Secchi disk transparency were the only variables that explained most of the variation in the data. The first analysis (Test 1, Table 7) proved good for separating out the seasonal data for Lake Gatlin, but showed poorer discriminating ability when trying to discriminate samples from the other pools, especially between the west and east pools and the south and middle pools (Table 7). This problem was due to the high degree of similarity of the data of the contiguous pools. A considerable improvement of the discriminant functions was obtained by combining observations from the pools of the same lakes (i.e., the east and west pools and the middle and south pools). The first two canonical variables from this final analysis (Test 4) can be used to cluster the data into three groups with some reliability as illustrated in Figure 4.

While condictivity was the best variable in terms of discriminating the lake pools, this variable reflected changes in the hydrologic regimes more than actual changes in lake trophic state. The discriminant analysis showed that Secchi disk transparency was almost as good as a discriminator by itself and also reflected changes in the trophic conditions in the system with some regularity. For this reason it was felt that the transparency alone could be used to indicate changes in the trophic state of the Lake Conway system.

ZOOPLANKTON

Seven species of copepods, 14 species of cladocerans, and 24 species of rotifer were identified in samples collected from the Lake Conway system. At any particular time, two to four species of copepods and cladocerans, and one to three species of rotifers were encountered in the lakes. Shannon-Weaver diversity indices computed for the individual samples were generally low, ranging from 0.80 to 2.46. However, low zooplankton diversity is characteristic of Florida lakes (Nordlie 1976) and North American lakes in general (Pennak 1962). All of the zooplankton species encountered during this study have been collected form other Florida lakes (Nordlie 1976; Confer 1971; Maslin 1969; Cowell *et al*. 1975).

The 24 month averages of areal abundances of the major taxa reveal some interesting trends (Table 8). Copepods comprised over 43 and 42 percent of total zooplankton in the south and middle pools, respectively, with cladocerans and rotifers of lower inportance. *Diaptomus floridanus* was the dominant organism in these two pools. Rotifers dominated the west pool while in the east pool all three of the major taxa were nearly codominant. Forty-three percent of the total zooplankton in Lake Gatlin were cladocerans, primarily *Bosmina longirostris*. Average annual abundance of total zooplankton ranged between 201,351 and 378,906 individuals per square meter of lake surface area (Table 8).

The results of the zooplankton study indicate several important trends that may have significance in the development of zooplankton communities in Florida lake systems. Total zooplankton abundance increased moving north

Table 6. RESULTS OF STEPWISE DISCRIMINANT ANALYSIS ON THE LAKE CONWAY TROPHIC INDICATORS.

Test	Factors Considered	Number of Groups	Variable Entered	F'	$F_{0.95}(\nu_1, \nu_2)$*
1	Conductivity, Secchi disk transparency, chlorophyll *a*, total nitrogen, total phosphorus, productivity	5	Conductivity Secchi disk	15.88 10.60	4.00 4.00
2	Conductivity, Secchi disk transparency, chlorophyll *a*, total phosphorus	5	Conductivity Secchi disk	15.86 10.60	4.00 4.00
3	Secchi disk transparency, chlorophyll *a*, total nitrogen, productivity	5	Secchi disk	10.26	4.00
4	Conductivity, Secchi disk transparency, chlorophyll *a*, total nitrogen, total phosphorus, productivity	3	Conductivity Secchi disk	32.11 21.34	4.00 4.00

*$\nu_1 = G - 1$; $\nu_2 = N \cdot G - V$ (G = No. of groups, N = Total no. of samples, V = No. of variables).

Table 7. DISCRIMINANT FUNCTIONS DEVELOPED FROM THE LAKE CONWAY TROPHIC INDICATOR DATA. THE CLASSIFICATION MATRIX SHOWS THE NUMBER OF SAMPLES PLACED IN EACH GROUP AND THE PERCENT OF CORRECT CLASSIFICATIONS.

Classification Functions

	Test 1					Test 4		
Group	South	Middle	East	West	Gatlin	South Middle	East-West	Gatlin
Variable								
Conductivity	0.311	0.313	0.339	0.346	0.409	0.317	0.349	0.416
Secchi disk	2.237	2.057	1.198	1.112	-0.177	2.194	1.189	-0.162
Constant	-39.457	-39.273	-42.686	-43.929	-56.946	-39.506	-43.506	-57.359

Group	Percent Correct	Number of Cases per Group				
		South	Middle	East	West	Gatlin
South	50.0	11	1	5	5	0
Middle	18.2	10	4	2	6	0
East	18.2	4	3	4	8	3
West	40.9	1	3	6	9	3
Gatlin	86.4	0	0	1	2	19
Total	42.7	26	11	18	30	25

Group	Percent Correct	No. of Cases per Group		
		South-Middle	East-West	Gatlin
South-Middle	59.1	26	18	0
East-West	56.8	11	25	8
Gatlin	86.4	0	3	19
Total	63.6	37	46	27

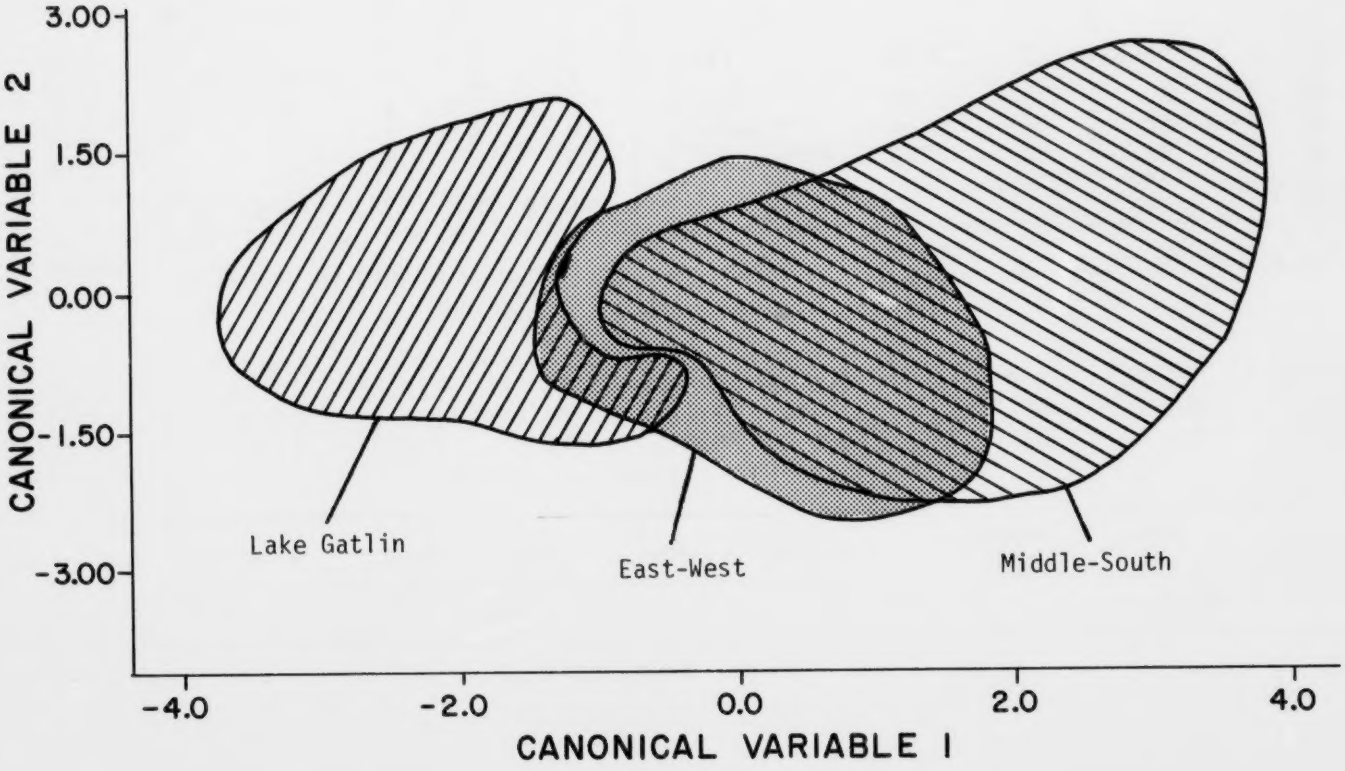


Figure 4. Canonical variables resulting from the discriminant analysis (test 4) on the six trophic state indicators.

Table 8. AVERAGE ABUNDANCE (24 MONTHS) OF MAJOR TAXA AND TOTAL ZOOPLANKTON IN THE VARIOUS POOLS OF THE LAKE CONWAY SYSTEM.

Pool	South	Middle	East	West	Gatlin
Total Copepods*	72130	68255	65614	55883	59552
Total Cladocerans	45870	37830	48161	55143	115017
Total Rotifers	43674	47383	64232	71724	86148
Total Zooplankton** (NO/m^2)	169056	161065	195454	199116	266960
Total Zooplankton** (NO/L)	28	27	43	42	64

*Includes Nauplii
**Includes Protozoans

through the pools, associated with increasing trophic state. This trend also has been observed for other lake systems (Patalas and Salki 1973). The increase in the Lake Conway system is attributable to an increase in the abundance of small cladocerans and rotifers, taxa which are indicative of eutrophic conditions.

INTERRELATIONSHIPS

Nitrogen and phosphorus stormwater loadings to the various pools varied concommitantly with watershed lake area ratios (Table 9). The South and middle pools of Lake Conway which had a watershed Lake area ratios of about 1.0 received a minor portion (< 10%) of its nutrient loadings from stormwater. Conversely Lake Gatlin with a A_w/A_m ratio of 11.18 received about half of its nutrient load from this source.

Table 10 compares nitrogen and phosphorus stormwater loadings to the water quality and biological indicators from the Lake Conway system. Significant correllations between the various parameters are presented in Table 11. Strong relationships were found between watershed - lake area ratios and stormwater nutrient loadings ($r^2 = 0.98$); between stormwater nutrient loadings and secchi disk visibility ($r^2 = 0.98$); between secchi depth and chlorophyll a ($r^2 = 0.93$) and total zooplankton ($r^2 = 0.98$); and between chlorophyll a and zooplankton ($r^2 = 0.88$). Significant relationships were indicated between zooplankton and watershed - lake area ratios and zooplankton and nitrogen stormwater loadings but these were considered to be spurious.

These results indicate that those pools of the Lake Conway system which received a larger share of their loadings from stormwater sources exhibited a degredation in water quality as indicated by changes in secchi disk visibilities and chlorophyll a. Lake Gatlin, which received about half of its nutrient loadings from stormwater had an average secchi disk visibility which would place the lake in the eutrophic category.

It is likely that in response to the increase in algal biomass (as indicated by chlorophyll a) in those pools that receive larger stormwater loads, zooplankton abundance also increased. Additionally, qualitative changes in the zooplankton communities also occurred. However, one must always be aware that other factors such as predation modify zooplankton communities and hence the response zooplankton exhibit toward increased nutrient loading.

CONCLUSIONS

A study of nutrient sources to the Lake Conway, Florida, Ecosystem has shown that inputs form stormwater runoff to each of the pools of this three lake system varied in proportion to the watershed lake area ratio. Total nutrient inputs to the lakes were within the reange of loadings that leads to mesotrophic conditions.

Multivariate analysis of water quality data by the discriminant analysis showed differences among the three lakes of the Conway System. Water quality and biological indicators were significantly correllated to inputs from stormwater runoff. These findings indicated a direct impact of stormwater runoff on water quality and subsequently on changes in the lakes biota.

Table 9. NITROGEN AND PHOSPHORUS STORMWATER LOADINGS FOR THE LAKE CONWAY SYSTEM NUMBERS IN PARENTHESES INDICATE PERCENTAGE OF TOTAL LOADING.

Lake	Lake Area (Km^2)	Watershed Area (Km^2)	[1] A_W/A_L	[2] L_n (g/m^2-yr)	[3] L_p (g/m^2-yr)
South Pool	1.37	1.75	1.28	0.13 (7.5)	0.015 (9.0)
Middle Pool	2.59	3.15	1.05	0.10 (6.8)	0.012 (7.6)
East Pool	1.27	9.92	7.81	1.11 (39.3)	0.134 (46.3)
West Pool	1.48	5.46	6.73	0.96 (38.1)	0.116 (44.1)
Lake Gatlin	0.28	3.13	11.18	1.94 (46.9)	0.232 (57.0)

[1] A_W/A_L = Watershed area to Lake area ratio.

[2] L_n = Stormwater nitrogen loading

[3] L_p = Stormwater phosphorus loading

Table 10. STORMWATER LOADINGS, WATER QUALITY INDICATORS AND BIOLOGICAL INDICATORS FOR THE VARIOUS POOLS OF THE CONWAY SYSTEM.

Lake	L_n (g/m^2-yr)	L_p (g/m^2-yr)	TN mg/l	TP mg/l	Secchi depth (m)	Chlorophyll a (mg/l)	Total Zooplankton (#/l)
South Pool	0.13	0.015	0.73	0.016	3.45	2.8	28
Middle Pool	0.10	0.012	0.77	0.018	3.32	3.7	27
East Pool	1.11	0.134	0.75	0.021	2.71	4.14	43
West Pool	0.96	0.116	0.75	0.020	2.66	4.47	42
Lake Gatlin	1.94	0.232	0.84	0.021	1.86	9.49	64

Table 11. COEFFICIENTS OF DETERMINATION (r^2) DERIVED FROM CORRELLATING THE VARIOUS LOADING PARAMETERS, WATER QUALITY INDICATORS AND BIOLOGICAL INDICATORS.

	A_w/A_L	L_p	L_n	TP	TN	Secchi Depth	Chlorophyll a	Total Zooplankton
L_p	0.98**	1.0						
L_n	0.98**	0.99**	1.0					
TP	--	--	--	1.0				
TN	--	--	--	--	1.0			
Secchi Depth	0.94*	0.98**	0.98**	--	--	1.0		
Chlorophyll a	--	--	--	--	--	0.93*	1.0	
Total Zooplankton	0.94*	--	0.98**	--	--	0.98**	-.88*	1.0

-- = $r^2 < 0.88$
*2 = 0.05
**2 = 0.01

ACKNOWLEDGEMENTS

I am expecially indebted to the U. S. Army Corps of Engineers Waterways Experiment station, Aquatic Plant Control Research Program, Vicksburg, Mississippi for its support and cooperation throughout this study. Special thanks are extended to Mr. Lewis Decell, Mr. William Rushing and Mr. Russel Theriot for their sustained interest in the project.

Acknowledgement is given to the Department of Environmental Engineering Sciences, Univeristy of Florida, and especially to Dr. Jackson L. Fox and Dr. Patrick L. Brezonik whose advise and guidance made this study possible. The assistance of Charles R. Fellows both in the field and laboratory was greatly appreciated.

The Assistance of the Marine Environmental Sciences Consortium, Dauphin Island, Alabama, and in particular the cooperation of Dr. George Crozier, director, Linda Lutz, illustrator and Pam Barbour and Lynn Bryant, typist, is gratefully acknowledged.

LITERATURE CITED

APHA. 1975. Standard methods for the analysis of water and wastewater, 14th ed. New York.

Blancher, E. C. 1979. Lake Conway, Florida, Ecosystem: Nutrient Dynamics, Trophic state. Zooplankton relationships. Ph.D. dissertation Univ. of Florida, Gainesville. 146 p.

__________, C. R. Fellows, D. Sompongse, and J. L. Fox. 1978. Nitrogen and phosphorus loading characteristics of the Lake Conway, Florida, ecosystem. Preliminary Rep. Dep. Env. Eng. Sci. Univ. of Florida, ecosystem. Preliminary Rep. Dep. Env. Eng. Sci. Univ. of Florida, Gainesville Fla. 47 p.

__________ and C. R. Fellows. 1979. Nitrogen and phosphorus loadings of the Lake Conway ecosystem: Final report. Final report to U. S. Army Engineer Waterways Experimental Station. Department of Environmental Engineering. University of Florida, Gainesville. 62 p.

Confer, J. L. 1971. Intrazooplankton predation by Mesocyclops edax at natural prey densities. Limnol. Oceanogr. 16: 663.

Conley, R. A., E. C. Blancher, F. S. Kooijman, and C. Feerich. 1978. The plankton, periphyton and benthic invertebrates of Lake Conway, Florida. Dept. of Environmental Engineering. University of Florida, Gainesville.

Cowell, B. C., C. W. Dye, and R. C. Adams. 1975. A synoptic study of the limnology of Lake Thonotosassa, Florida. Part I. Effects of primary treated sewage and citrus wastes. Hydrobiologia 46: 301.

Dixon, W. J. and J. Brown. 1977. BMDP-77, Biomedical computer programs, P-series. University of California Press. Berkeley. 880 p.

Edmondson, W. T. 1972. Nutrients and phytoplankton in Lake Washington. In G. E. Likens. ed. Nutrient and Eutrophication: the limiting nutrient controversy. Special Symposium, Amer. Soc. Limnol. Oceanogr. 1: 172-193.

Ewel, K. C. and T. D. Fontaine, III. 1977. Proposed relationships between white amur and aquatic ecosystem at Lake Conway, Florida. U. S. Army Engineer Waterways Experiment Station. M. P. A-77-3: 159-176.

Fellows, C. R. 1978. The significance of seepage in the water and nutrient budgets of selected Florida lakes. M. S. Thesis. College of Engineering University of Florida, Gainesville.

Fontaine, T. D. and K. C. Ewell. 1978. Baseline studies for evaluating the response of an ecosystem to the introduction of white amur. Second Annual Report to the U. S. Army Waterways Experiment Station. Center for Wetlands. University of Florida, Gainesville (mimeo).

Fox, J. L., E. C. Blancher, F. M. Kooijman, R. A. Conley, and C. P. Feerick. 1977. Biological baseline studies of the Lake Conway, Florida, system. U. S. Army Engineer Waterways Experiment Station. M. P. A-77-3: 123-145.

Gakstatter, J. M., M. O. Allum, and J. M. Omernik. 1975. Lake eutrophication: Results from the National Eutrophication Survey. Corvallis, Ore. 32 p.

Guillory, V., R. Land, and R. Gasaway. 1977. Baseline data report -- Lake Conway grass carp project. Fishes. U. S. Army Engineer Waterways Experiment Station. M. P. A-77-3:36-112.

Kluesener, J. W. and G. F. Lee. 1974. Nutrient loading from a separate storm sewer in Madison, Wisconsin. J. Water Poll. Contr. Fed. 46: 921 936.

Lee, D. R. 1977. A device for measuring seepage flux in lakes and estuaries. Limnol. Oceanogr. 22: 140-147.

Maslin, P. E. 1969. Population dynamics and productivity of zooplankton in two sandhills lakes. Ph.D. Dissertation. University of Florida, Gainesville.

Mattraw, H. C. and C. B. Sherwood. 1977. Quality of storm-water runoff from a residential area, Broward County, Florida. Jour. Research U. S. Geol. Survey 5: 823-834.

Nall, L. E., M. J. Mahler, and J. Schardt. 1977. Aquatic macrophyte sampling in Lake Conway. U. S. Army Engineer Waterways Experiment Station. M. P. A-77-3:113-122.

__________, J. D. Schardt, and A. P. Burkhalter. 1978. The effect of the white amur on the aquatic vegetation of Lake Conway, Florida. First Annual Report, Baseline Data. Florida Department of Natural Resources. Tallahassee, Florida. 161 p.

NOAA. 1972-1978. U. S. Climatological reports. Florida section. National Oceanographic and Atmospheric Agency, Washington, D. C.

Nordlie, F. G. 1976. Plankton communities of three central Florida lakes. Hydrobiologia 48: 65.

__________ and A. Salki. 1973. Crustacean plankton and the eutrophication of lakes in the Okanagan Valley, British Columbia. J. Fish. Res. Bd. Can. 30: 519.

Pennak, R. N. 1962. Species composition of limnetic zooplankton communities. Limnol. Oceanogr. 2:222.

Sawicki, A. T. 1977. Background water quality analysis of Lake Conway. U. S. Army Engineer Waterways Experiment Station. M. P. A-77-3: 146-158.

Sompongse, D. 1978. A study of the nitrogen cycle in the Lake Conway, Florida, ecosystem. M. S. Thesis. College of Engineering, University of Florida, Gainesville.

__________. 1976. Methods for chemical analysis of water and wastes. Environmental monitoring and support laboratory, Cincinnati, Ohio. 298 p.

Weibel, S. R. 1969. Urban drainage as a factor in eutrophication. In: Eutrophication: Causes, consequences and correctives. Natl. Acad. of Sci. Washington, D. C. 383 p.

THE DISTRIBUTION OF SEDIMENTS AND PARTICULATE CONTAMINANTS FROM COMBINED SEWER AND STORM DRAIN OVERFLOWS IN SEATTLE'S NEARSHORE WATERS

R.D. Tomlinson, B.N. Bebee, and R.G. Swartz
Municipality of Metropolitan Seattle, Seattle, Washington
S. Lazoff and D.E. Spyridakis
University of Washington, Seattle, Washington

ABSTRACT

The distributions of particulates and the associated contaminants emanating from combined sewer outfalls and storm drains in Lake Washington were examined. In a preliminary study sediment samples were collected by SCUBA divers from the area of greatest apparent contamination at each of 20 outfalls. The sample content of organics, heavy metals, oils and greases, and pesticides was used to select stations for more intensive study. At each of those sites quality and quantity analyses of the overflows provided loading estimates for the wastewater particulates entering the nearshore waters. Light transmission measurements of storm-induced wastewater plumes helped to define the nature of plume dispersion and its area of significant influence. Organic carbon and heavy metals analyses of settling particulates and surface sediments near the outfalls supplied further detail and confirmation of the fate of the wastewater particulates.

ACKNOWLEDGEMENTS

This work was done by personnel of the Municipality of Metropolitan Seattle and the University of Washington Department of Civil Engineering under a grant from the U.S. Environmental Protection Agency. The authors gratefully acknowledge this support.

INTRODUCTION

There are about 110 overflow locations in the Seattle area, of which 30 are the Municipality of Metropolitan Seattle's (Metro's) and the remainder are the responsibility of the City of Seattle. A brief, intensive rainstorm can gorge city collector sewers, causing basement backups and discharges to Lake Washington. In contrast, a milder storm of longer duration will tend to flow more smoothly to the larger trunk sewers where a buildup of flow beyond treatment and interceptor trunk capacity will lead to programmed overflows to the lower Duwamish River, Elliott Bay and Lake Union. These longer storms, typical of the winter season, may result in less than 50% of sewer wastewater actually reaching the West Point Treatment Plant during a given storm period. Operation of Metro's Computer Augmented Treatment and Disposal (CATAD) System, which provides in-line storage in some of the larger sewers, permits some flexibility as to discharge location, with highest priority given to protection of freshwater bodies (Lake Washington, Lake Union) where possible. Frequency of overflow at present is estimated to average about 40 occurrences per outfall per year, of which perhaps five or six are summer occurrences.

Combined sewer and storm drain outfalls are scattered throughout the Seattle area, discharging varying volumes to all of the major water bodies around the city. Some of these discharges are over shellfish beds, or as in Lake Washington, near public bathing beaches or spawning areas for anadromous fish; others are in the harbor area where water contact sports or potential public health hazards are minimal.

Summer season dye studies done in 1976 indicated that the nearshore surface circulation at two lake stations was comparatively sluggish, providing a maximum dilution of 2.7:1 (Lake Washington) and 75:1 (Lake Union) during the first three hours following release. Such conditions might be expected to favor the nearshore settling of much of the sediment injected into the environment by combined sewer overflows and storm drains. Very little information has been available concerning the dispersion of these materials.

To help fill the gaps in our knowledge of the distribution and fate of wastewater particulates in Seattles's nearshore waters, EPA and Seattle Metro cosponsored a research project with the following objectives:

1. To determine the fate and distribution patterns of suspended particulates emanating from representative combined sewer overflows and storm drains in the Seattle area through the correlation of quantitative in-situ observations with suspended solids loading and current patterns.

2. To determine the distribution patterns and ultimate fate of selected particulate wastewater contaminants, including copper, lead, zinc, carbon and phosphorus.

METHODS AND MATERIALS

EFFLUENT

Quantity and Quality

For selected rainstorms the overflow effluents at a representative combined sewer outfall and storm drain (CSO 023 and SD 7 - refer to Figure 2 for locations) were sampled for loading analysis of suspended solids, metals, total organic carbon, total phosphorus, oils and greases, and total chlorinated hydrocarbons. Automatic sequential samplers triggered by pressure switches were used in conjunction with Arkon pressure-driven flow recorders to provide 2-liter samples at regular intervals during the overflows. The sampling intervals were chosen according to the speed of flow response in each system, so that samples would be taken that were representative of all the major high and low points on the overflow hydrographs.

Individual analyses were performed as follows:

Total suspended solids - Determined in conjunction with the total chlorinated hydrocarbons analyses, as specified below.

Copper, lead, zinc, aluminum and mercury - Split sample, acidified one part with conc. HNO_3. Filtered one half of sample through a .45μ membrane filter prerinsed with 1% HNO_3 and deionized water. Acidified filtrate and unfiltered aliquot sample with conc. HNO_3; digested the latter. Both portions analyzed for Cu, Pb, Zn, Al by AAS, and for Hg by cold vapor flameless AAS.

Total phosphorus - Split sample, filtered one part through Whatman #40 filter. Acidified samples were digested as for Cu, Pb, and Zn. PO_4 measured as orthophosphate by the ascorbic acid method (APHA, 1975).

Chlorinated hydrocarbons - Analyzed particulates only. Filtered sample through muffled, tare-weighed Whatman GF/A glass fiber filters. Filter dried at 105°C, cooled, weighed and Soxhlet extracted with acetone for eight hours. Extracts dried, infused with petroleum ether and concentrated. Cleanup and fractionation of dissolved components done in Florisil columns by elution with various mixtures of ethyl and petroleum ether. Analyses done by GC, with a Tracor Model 222 GC fitted with a ^{63}Ni, high temperature, electron capture detector.

Oils and greases - Hexane/Soxhlet extraction method (APHA, 1975).

Total organic carbon - $K_2S_2O_8/H_3PO_4$ digestion of blended samples and standards in nitrogen-bubbled, sealed ampoules. Resultant CO_2 quantified using a Model 865 Beckmann Infrared Analyzer.

Estimates of total loading for each measured parameter for each outfall for each storm were made with the aid of a computer program designed to calculate a time integral product of concentration times flow for each storm hydrograph. To this end the flow data were entered as values measured at regular intervals throughout. The quality analyses were less abundant, being limited specifically to the significant features of the hydrograph, i.e., the highs, the lows and the initial and final ascending and descending limbs. Between these points, values were interpolated to complement the flows given at regular intervals. A storm was defined as a minimum total of .03 in. of rain separated from preceding and succeeding measurable rainfall by a minimum of three hours.

Single quality samples contained insufficient particulate material for chlorinated hydrocarbon analyses. Therefore, sequential samples were composited to represent specific features on the hydrographs. With the exception of the final storm monitored at each of the two principal outfalls, the sediment loading values were likewise calculated from storm segment composites.

For the storm drain, where a dry weather base flow is discharged, pollutant mass loadings were calculated for both the base flow and the storm washoff--the latter being determined as the difference between the total storm load and the base load estimated from preceding flows and concentrations. Total rainfall volume on each service area was also determined.

Plume Distributions

During a given storm a grid of up to 24 stations was covered at each sample site; at each grid point a Martek Model XAS Transmissometer was lowered to the bottom, giving a real-time plot of percent light transmission versus depth on a shipboard XY plotter. Each complete grid coverage required about one hour, and was repeated whenever possible in order to measure various stages of the plume development and dispersion.

For grid-site location, a one-time survey was carried out wherein onshore-offshore transects were established visually using landmarks. Rangefinder distances were then determined along these transects using buoy markers when needed. Depth readings from the shipboard depth-finder were recorded for each point and used for all subsequent outings to determine station locations along the transect. The estimated maximum positional error for the stations farthest offshore was 15 m and was somewhat less for inshore stations.

Mounted on the transmissometer sensor, temperature and oxygen probes were used to periodically check the stratification of the water column. These sensors were interfaced to a Martek Mark V Water Quality Monitor and an XY plotter. Both units were lab-calibrated prior to use in the field.

The resultant vertical profiles of light transmission versus depth were digitized and keypunched for computer contouring. For contouring purposes an interpolative contouring routine was used to fit a bivariate elastic surface to the initially coarse, raw data grid, resulting in smooth contour output. This program was a modified version of a program described in Numerical Plotting System User's Manual No. W00053 (University of Washington; August, 1977).

SETTLING PARTICULATES

Settling particulate matter was sampled continuously from 1/30/78 to 1/29/79 near the wastewater outfalls at CSO 023 and SD 7, and from 10 to 60 m offshore at the Control 3 site (Figure 2). The sampling devices, or sediment traps, consisted of 30 cm X 30 cm polyvinyl chloride platforms, which held four 10-cm diameter funnels joined to 50-ml centrifuge tubes (Figure 1). All parts of the platforms (including screws) were made of plastic to limit metals contamination. The collection surface for each trap was suspended 2 m above the lake bottom. For each collection period the traps were serviced and cleaned, with retrieval and reset being done by SCUBA divers to minimize agitation of bottom sediments.

Arrays of six sediment traps were moored at each station, with each array covering approximately .5 acre (Figure 4). Sediment trap placement was determined by visual and photographic observations of the divers. The rather distinct line of transition between the effluent debris and the ambient sediments was denoted the plume boundary.

The sediment trap samples were centrifuged at 9000 rpm for 7 minutes, decanted and dried at 60°C for 36 hours. The samples were then weighed individually and the four samples for each trap were combined and comminuted prior to chemical analysis. One hundred mg subsamples were then digested, using $HF/HNO_3/HClO_4$ (Bortleson and Lee, 1972) in a 10-ml teflon crucible (Birch, 1976). An aliquot of this digestrate was used for determination of P, Cu, Pb, Zn and Al.

The analyses were done as follows:

Phosphorus - Ascorbic acid molybdenum blue method (APHA, 1975). The mean of duplicates was reported for each sample.

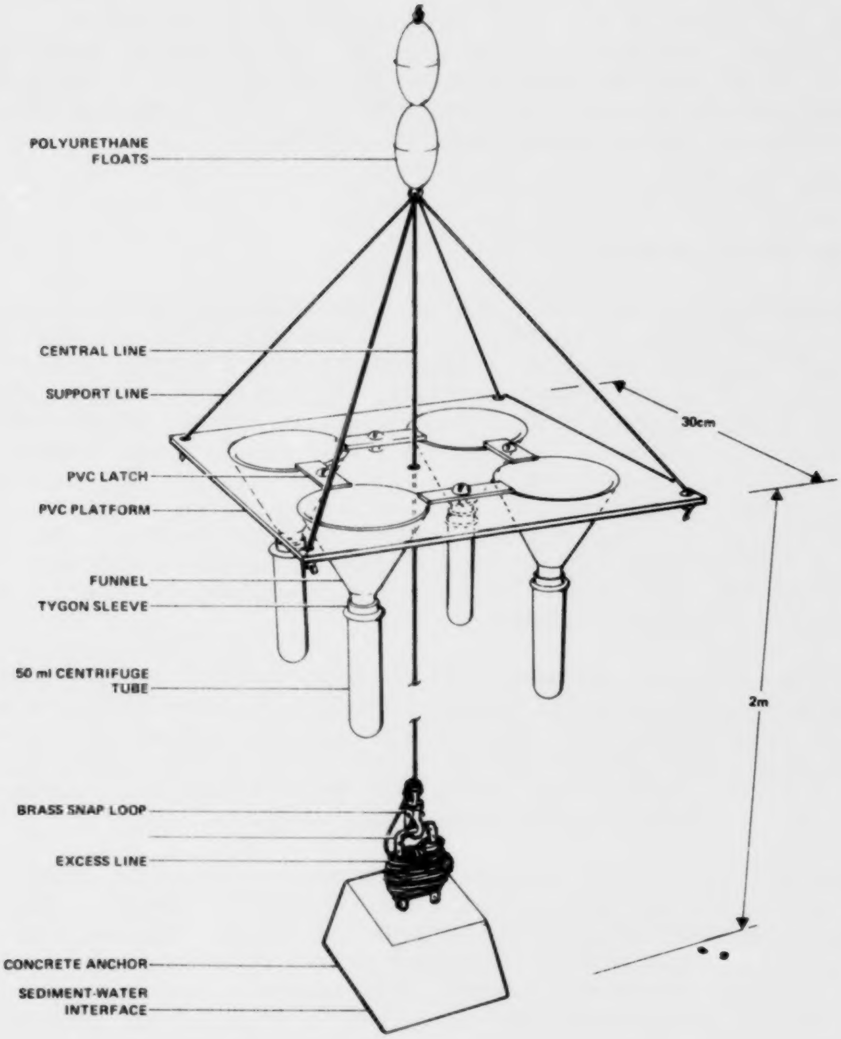


Figure 1. Details of sediment trap configuration.

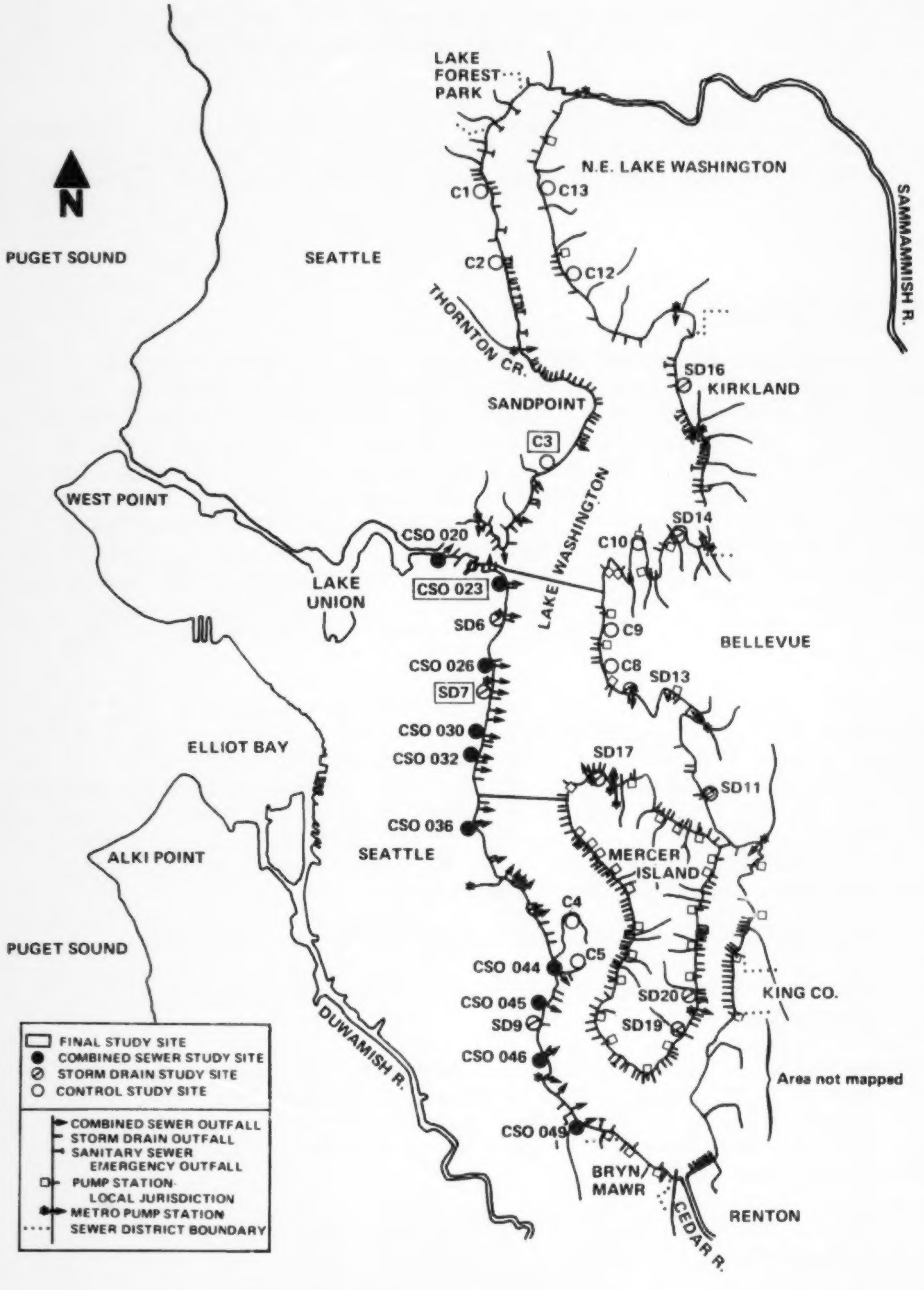


Figure 2. Locations of sampling sites.

Copper, lead, zinc and aluminum - Digestrate analyzed by AAS. Al also analyzed by neutron activation.

Total carbon - Analyzed 20 to 100 mg of dried sediment using a Leco induction furnace and carbon analyzer. Results reported as the mean of duplicate analyses.

SEDIMENTS

Surface Samples

Sampling for preliminary analyses of the freshwater sediments was carried out by SCUBA divers. Each station was initially surveyed as to depth and bottom contours, nature and condition of the outfall pipe, extent of apparent debris accumulation, and proximity and nature of intefering structures. At each CSO and SD, duplicate samples of the surface 3 cm of the (apparently) most highly contaminated sediments were collected in specially prepared 110-cm^3, glass cylinders closed at each end with rubber stoppers. The stoppers of one of each set were covered with aluminum foil to facilitate pesticide/PCB analyses. For the aluminum analyses the stoppers of the other were left bare.

The excess water was drained off each of the surface sediment samples collected for the preliminary study, and the sediment was transferred to glass containers for PCB and total chlorinated hydrocarbon analysis and polypropylene containers for all other analyses. The samples were weighed wet and then oven-dried at 105°C. Aliquots taken for Hg analysis were dried at 50°C to minimize loss through vaporization. All samples were weighed dry when cool.

Individual analyses were performed as follows:

Copper, lead and zinc - Analysis as for settling particulates.

Mercury - Dried sediments digested for one hour in conc. H_2SO_4/conc. HNO_3 mixture, infused with a $KMnO_3/K_2S_2O_8$ mixture and digested for an additional two hours. Analyses were done by cold vapor, flameless AAS.

Total phosphorus - Digestion and analysis as for effluent.

PCB/chlorinated hydrocarbons - Wet sediments extracted and analyzed as for effluent.

Oils and greases - Hexane/Soxhlet extraction method (APHA, 1975).

Total organic carbon - Preliminary study: inorganic carbon driven off as CO_2 by acidification of samples to pH = 2.0 with HCl. Residual (organic) carbon measured as combustion CO_2 by a Leco Model WR-11 Total Carbon Analyzer. Intensive studies: as for settling particulates.

Cores

Lake sediment cores for analysis of sediment profiles of metals and carbon were also collected by divers at the transmissometer grid sites, using 30 cm lengths of clear tenite butyrate tubing (3.5 cm ID). Cores from depths greater than 15 m were taken with a 7-kg modified Phleger gravity corer.

The cores were examined for gross physical characteristics and then extruded and sliced at 0.5 cm and at 1-cm intervals to 12 cm depth. The slices were weighed, dried at 103°C for 24 hours, reweighed and ground in a Diamonite mortar. Digestion and analysis were done as described for settling particulates on the 0-0.5 cm, 0.5-1 cm, 3-4 cm and 7-8 cm sections. Analysis for chlorinated hydrocarbons was as for effluent. Some cores were stored at 5°C for up to one week before processing.

RESULTS AND DISCUSSION

PRELIMINARY STUDIES

Selection of Sampling Sites

Prior to the present study, no comprehensive map was available giving the locations and dimensions of the sewer outfalls in Lake Washington. Drawing information from a number of sources, we compiled as much of the available outfall information as feasible to provide a comprehensive choice of study sites. A base map constructed from subbasin drainage charts (RIBCO, 1974) was augmented with further details obtained through visits to the files of agencies representing the nine sewer districts surrounding the lake. The resultant map (Figure 2) included 23 emergency outfalls (activated only by power failure), 34 CSOs, 56 pump stations and 240 SDs. For Lake Washington's 71.5 miles of shoreline, this gave a conservative average of five outfalls per mile. The map, although incomplete, was felt to include all of the major drainage systems.

On the basis of the compiled information and further details obtained through field outings, sampling sites (10 CSOs, 10 SDs and 10 control locations) were selected for the preliminary study (Figure 2). In this, we were guided by three principal criteria: 1) that the locations be as free as possible of contamination from other outfalls; 2) that potential difficulties with monitoring equipment installation, servicing and protection be minimal; and 3) that the selected facilities have a high probability of large and frequent storm overflows or direct drainage responses.

Outfall interferences were judged from map information. Data on estimated (modelled) overflow frequency, volume and duration for each of the CSOs were obtained from the City of Seattle Engineering Department; because similar information was unavailable from the SDs, those facilities having the largest drainage basins were chosen for study.

Relative Levels of Sediment Enrichment

At each of the CSO, SD and control sites selected, samples of the surface (3 cm) layer of sediments were collected by divers at the near-outfall locations thought to have the maximum potential for contamination by the wastewater effluents. These samples were analyzed for Cu, Pb, Zn, Hg, total P, TOC, oils and greases, PCBs and chlorinated hydrocarbons.

Figure 3 represents an attempt to composite the resultant information. The "sediment enrichment factors" are given for three categories of parameters: 1) organics (TOC plus oils and greases) and total phosphorus, 2) metals (Cu, Pb, Zn, Hg), and 3) chlorinated hydrocarbons and PCBs. To calculate these relationships, mean values were derived for each parameter at each station and the resultant figures were normalized by parameter, i.e., the lowest value found for each parameter was divided into each of the corresponding values from the other 28 stations. These figures were then added for each station for each category, and the sums were in turn normalized within the categories. Thus, the total length of the three histogram bars at each station, divided by 3.0, gives the overall enrichment of that station compared to a hypothetical composite of the most pristine sediments found in the lake. The sediments from CSO 046, which does not appear to overflow, approach this quality overall. The background concentration standards used for the Figure 3 calculations were all comparable to or lower than those determined for pre-1900 core segments collected in the deep portion of the lake by Birch (1976) and Spyridakis and Barnes (1976).

Sediments around the sewer outfalls on the eastern shore of Lake Washington (where there are no combined sewers) generally appeared to be only one-third to one-fourth as con-

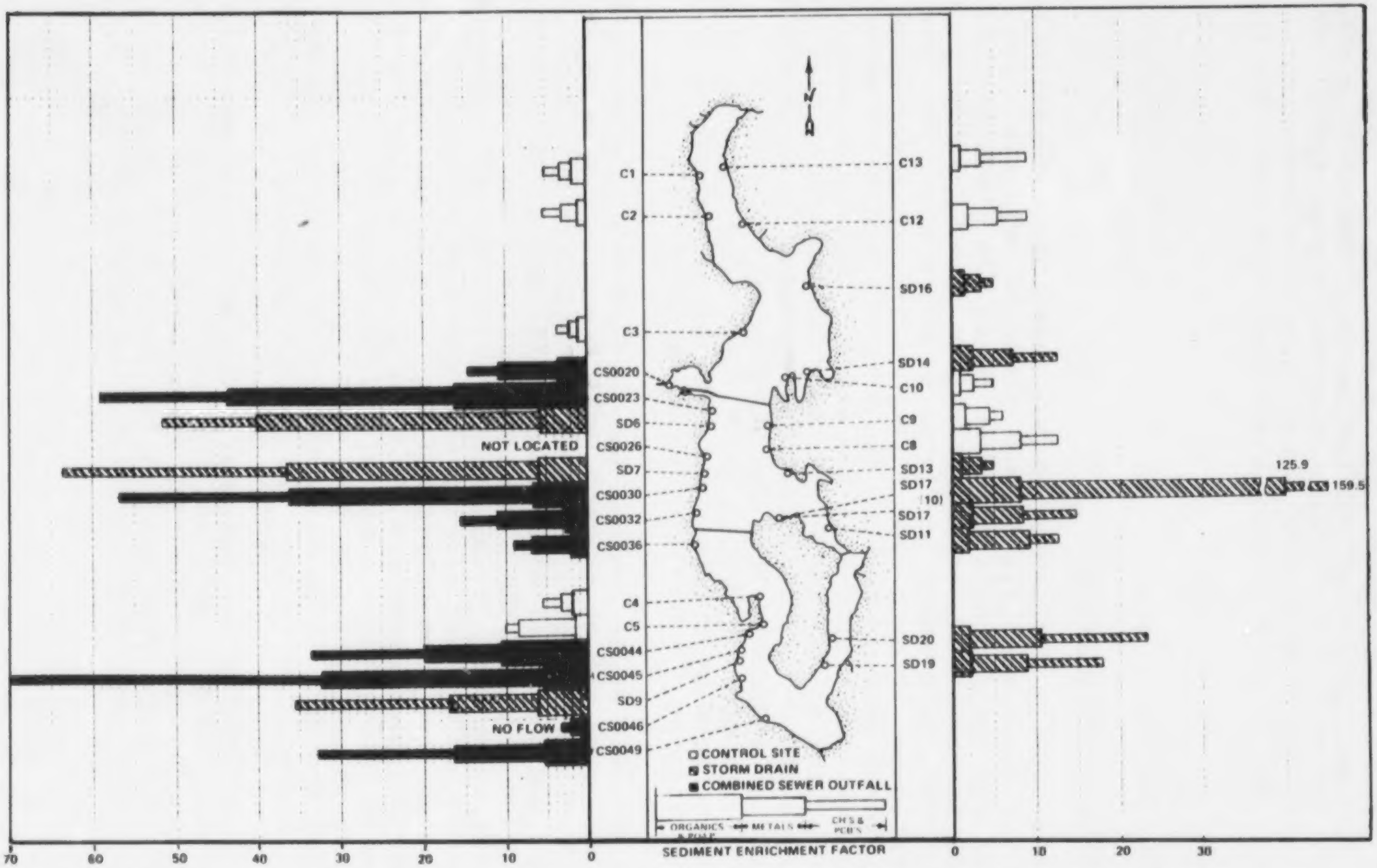


Figure 3. Relative enrichment of the nearshore surface sediments of Lake Washington with normalized categories of parameters (refer to text for details of data reduction). Organics and PO_4-P: total organic carbon, oils and greases, total phosphorus; metals: copper, lead, zinc, mercury; chlorinated hydrocarbons and PCBs: BCH, Lindane, Heptachlor, Heptachlor E, Aldrin, Dieldrin, Endrin, DDT (DDD + DDE + p,p DDT), Arochlor 1242, Arochlor 1248, Arochlor 1254, Arochlor 1260.

taminated overall as those between the two floating bridges and south of Seward Park (denoted by C 4 and C 5) on the western shore. Sediment pollutants at several stations in this area were enriched more than 16 times over background levels. The area between the bridges had particularly high concentrations of metals, which are typically associated with both CSOs and SDs.

INTENSIVE STUDIES

Description of Study Sites

The locations ultimately chosen were the best available but did not fully meet all of our specified criteria. For either of the two outfalls selected there was a substantial potential for effluent interference from other, nearby systems. This is evident in Figure 4, which shows the site configurations. In each case a large storm drain was found within several hundred feet of the sample outfall. However, the approximate dimensions of the zones of debris accumulation observed by the divers indicated that the separation distances might be sufficient to minimize interferences; also, probable current patterns deduced from dye studies in the general vicinity of the CSO (CH2M/Hill, 1974) indicated that the wastewater plumes there were likely to be parallel in most instances and not convergent.

The combined sewer outfall (designated CSO 023) was found to have a scoured pit at the end, with a built-up pile of debris somewhat further offshore. The pit was approximately half a meter deep and one meter wide, whereas the mound was about 1.5 m high and 14 m across. The discharge point for the principal storm drain system (SD 7) was nearshore in localized riprap and opened onto much steeper terrain than did that of the CSO, thus minimizing scouring and debris buildup. The outfall structures, which both had pipe diameters of 24 inches, were also found to have transient contiguous areas of visible debris accumulation as large as .5 acre, as indicated in Figure 4. Such evidence of scouring and deposition was found to be common at the many outfalls investigated for the preliminary studies.

The locations of the sediment traps and transmissometer grid sites at CSO 023, SD 7 and the control station (C 3) are also shown in Figure 4. The typical substrate in the various areas was: CSO 023 - fine sand, silt, light debris; SD 7 - fine sand, silt, some clay, light debris; control site (C 3) - sand.

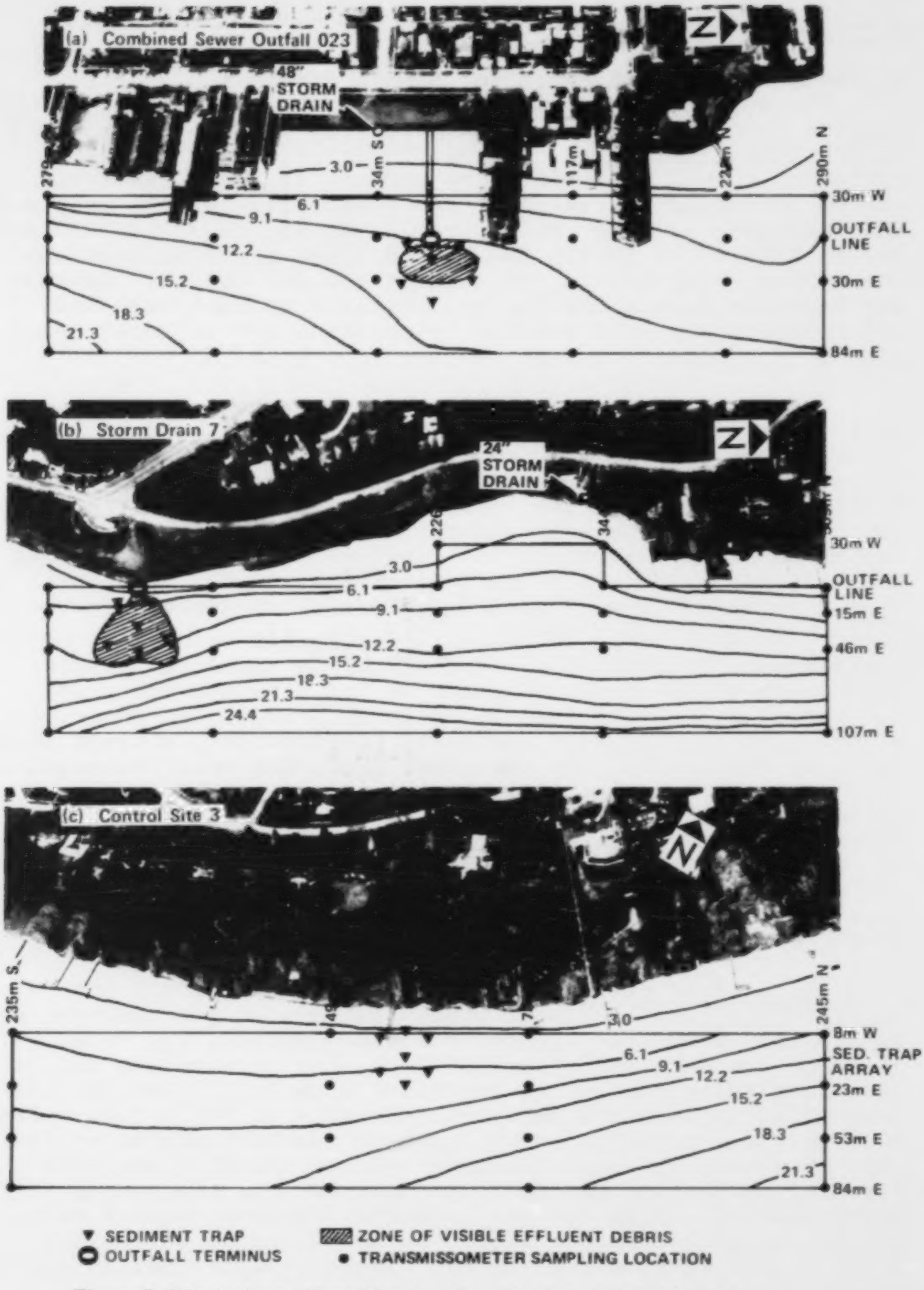


Figure 4. Principal set of intensive sampling sites in Lake Washington.

Overflow Monitoring

Effluent Loading Estimates for Single Events--

The wastewater effluents generated by several storms were sampled quantitatively and qualitatively at CSO 023 and SD 7 to determine typical particulate inputs to the freshwater environment by a representative CSO and SD. For the five storms monitored the rainfall range was 0.17 to 1.05 in., and the discharge volumes varied from 51 to 951 m^3 at CSO 023 and 213 to 1849 m^3 at SD 7. The months represented include the end of the rainy season and most of the dry season. Because the drainage basins for the two stations sampled were only 1.5 miles apart, with similar orientation and no intervening ridges, a single rainguage was used for both.

The similarities in relative size of the two drainage basins (CSO 023 - 117 acres, SD 7 - 125 acres), and in incident rainfall for any given storm, invited comparisons of their pollutant loading and runoff characteristics. The major distinction affecting the relative quality of runoff for these two areas is land-use distribution; the CSO 023 basin is comprised of approximately 75% single-family residences with the rest multiple-family residences, whereas the SD 7 basin has a significantly higher fraction of single residences (normally associated with lower pollutant accumulation rates than multiple residences). For three of four storms an appreciably lower percentage of the total rainfall on its drainage basin was discharged by CSO 023 than by SD 7; this was to be expected since a CSO discharge is typically intermittent, whereas that of an SD is continuous. On the other hand, the relative magnitudes of the mean storm discharge concentrations for the various pollutants varied from one storm to the next, as documented below. A statistical summary of the contaminant mass discharges is offered in Table 1. Mean ratios (SD 7:CSO 023) of per-storm mean concentrations and total loadings are also presented here for each parameter, in support of the ensuing discussion (Table 2).

Particulate effluent--With reference to Table 2 the SD 7:CSO 023 ratios of mean concentrations in particulates were appreciably less than 1.0 for suspended solids, Cu, total P, oils and greases, and chlorinated HC; the particulate ratios were near unity for Zn, Al and total organic C. Only Pb had a significantly higher solids concentration at SD 7, this being evidently due to dilution by low-lead sanitary wastes at the CSO. However, the mean discharge volume for SD 7 was over three times that of CSO 023; consequently the mean storm particulate loadings for all parameters except total P were greater at SD 7 than CSO 023. The mean loading values for total P were similar for the two sites, giving a SD:CSO ratio near 1.0.

TABLE 1. STATISTICAL SUMMARY OF ESTIMATED POLLUTANT LOADS AND CONCENTRATIONS IN STORM DISCHARGES MONITORED AT STORM DRAIN 7 AND COMBINED SEWER OUTFALL 023, MARCH–SEPTEMBER, 1978

	CSO 023		SD 7	
MASS RANGE (kg)		No. Storms		No. Storms
Suspended Solids	6.40 - 142	(4)	27.4 - 142	(5)
Total Cu	.001 - .039	(4)	.020 - .049	(5)
Total Hg	.0000 - .0008	(4)	.0000* - .0004	(5)
Total Pb	.001 - .041	(4)	.098 - .316	(5)
Total Zn	.003 - .113	(4)	.044 - .156	(5)
Total Al	.059 - 2.56	(3)	.762 - 6.62	(4)
Total Org C	.865 - 18.0	(2)	11.0 - 16.2	(2)
Total P	.059 - 1.28	(4)	.120 - .417	(4)
Total O & G	.584 - 9.41	(4)	.740 - 8.06	(5)
Partic Chlor HC	.316 - 100 mg	(4)	1.02 - 86.1	(5)
PARTICULATE MASS (% of Total)	$\bar{x}^{(1)}$ ± 1σ	No. Storms	$\bar{x}$ ± 1σ	No. Storms
Suspended Solids	100.0	(4)	100.0	(5)
Cu	78.1 ± 10.2	(4)	64.0 ± 8.7	(5)
Hg	88.4 ± 6.9	(2)	ND*	(5)
Pb	69.1 ± 14.2	(4)	88.1 ± 2.7	(5)
Zn	68.6 ± 19.6	(4)	64.1 ± 5.0	(5)
Al	96.0 ± 2.0	(3)	96.6 ± 0.9	(4)
Org C	42.1 ± 22.9	(2)	56.0 ± 8.1	(2)
P	28.5 ± 10.3	(4)	52.7 ± 13.1	(4)
O & G	NA		NA	
Chlor HC	ND		ND	
DISCHARGE CONC (ppm)	$\bar{x}^{(2)}$	Range of 1σ	$\bar{x}$	Range of 1σ
Suspended Solids	153	90.8 - 258	80.8	43.8 - 149
Total Cu	.052	.026 - .105	.042	.026 - .068
Total Hg	.0010	.0003 - .0020	ND*	ND
Total Pb	.057	.025 - .130	.218	.127 - .372
Total Zn	.125	.061 - .258	.099	.066 - .150
Total Al	2.58	1.18 - 5.65	3.51	2.83 - 4.36
Total Org C	17.9	16.6 - 19.3	10.8	8.09 - 14.4
Total P	1.76	1.05 - 2.94	.315	.153 - .648
Total O & G	12.9	6.71 - 25.0	4.08	2.94 - 5.66
Partic Chlor HC	.018 ppb	.004 - .081	.014 ppb	.006 - .034

(1) Arithmetic mean
(2) Geometric mean

NA Not applicable
ND Not determined
* Below detection limits

TABLE 2. INTERSTATION COMPARISON FOR STORM DRAIN 7 AND COMBINED SEWER OUTFALL 023 OF MEAN EFFLUENT CONCENTRATIONS AND POLLUTANT LOADINGS ESTIMATED FOR FOUR STORMS (1) MONITORED AT BOTH SITES

		Geom Mean of SD 7:CSO 023 Ratios(2)	
Parameter	Fraction	Mean Conc	Pollutant Load
Susp Solids	Total	0.6	1.8
Cu	Partic	0.6	2.0
	Total	0.8	2.7
Hg	Partic	*	*
	Total	0.8	0.6
Pb	Partic	5.1	16.1
	Total	3.9	12.5
Zn	Partic	0.8	2.6
	Total	0.8	2.7
Al	Partic	1.2	6.3
	Total	1.2	6.3
Total Organic C	Partic	0.9	5.0
	Total	0.6	3.4
Total P	Partic	0.4	1.1
	Total	0.2	0.6
Oils & Greases	Partic	NA	NA
	Total	0.3	0.9
Chlorinated HC	Partic	0.7	2.4
	Total	ND	ND

Discharge Volume 3.3

---

(1) Storms of 3/23/78, 3/24/78, 4/15/78, 8/31/78
(2) Ratios initially calculated for each storm
* Below detection limits
NA Not applicable
ND Not determined

There were consistently higher mass loadings of Al from SD 7 than from CSO 023, indicating a greater input of inorganic terrigenous materials to the lake by the storm drain. There was a large variation from one storm to the next in the mean concentrations and total mass loading of chlorinated hydrocarbons at both sites (Table 1). Trends at the two stations were similar in that DDT accounted for a very large portion (approximately 75%-95%) of the total loading of particulate pesticides during the spring storms, and only about half of the August-September storm total, which was itself appreciably lower than that of the other storms. Lindane was the only other pesticide consistently present at both stations. The persistent occurrence of DDT and lindane conforms to the observations of Brenner et al. (1978) for runoff from the Juanita Creek watershed in the northeastern segment of the Lake Washington basin. The occurrence of these contaminants in both the CSO and SD systems indicates that they were present in the storm drainage, rather than in the sanitary wastes.

Total effluent--The largest concentration discrepancies between the two stations were for total suspended solids, Pb and those parameters related to domestic wastes, i.e., total organic carbon, total phosphorus, and oils and greases; SD 7 had substantially lower mean total effluent concentrations than CSO 023 for all of these except Pb, which was appreciably higher there than at the CSO. Only Hg and total P had lower mean total pollutant loadings at SD 7 than at CSO 023.

Annual Rainfall, Flow and Loading Summary--

The total rainfall on the Seattle central business district for the March, 1978-February, 1979 study period was 38.9 in., a value near the annual average of 34.1 in. (Phillips, 1968). However, the total rainfall recorded for the study area on the western shore of Lake Washington was only 23.2 in., a value confirmed by three separate gauges. It is probable that this difference represents a shielding effect, or rain-shadow, with the hills of the city blocking part of the precipitation from the storm fronts, which typically come from the southwest. Lacking long-term, confirming data, it was assumed that this effect is typical for the study area, and that the rainfall measured for the study period was near the corresponding annual average. The 1978 spring and summer were somewhat wetter than usual, but the 1978-1979 winter was drier, compensating for most of the observed difference. For a total of 62 periods of measurable rainfall (separated by one or more dry days) the fraction of those storms resulting in increased discharge at one or both outfall stations was 76%; the average total rainfall for those storms that failed to generate runoff effluent was .04 in. The total discharge volumes for the 12-month monitoring period were: CSO 023 - 25,600 m^3 (6.76 MG); SD 7 - 27,700 m^3 (7.34 MG), + 50,200 m^3 (13.3 MG) baseflow. These values were used with the mean discharge concentrations given in Table 1 and with concentrations measured for nonstorm discharge at SD 7 to calculate total annual loads for the various parameters. The results are presented in Table 3.

TABLE 3. ESTIMATED TOTAL MASS OF SELECTED CONSTITUENTS IN EFFLUENT DISCHARGED BY COMBINED SEWER OUTFALL 023 AND STORM DRAIN 7 MARCH 3, 1978 TO FEBRUARY 28, 1979

	CSO 023		SD 7			
	Storm		Storm		Non-Storm	
	Total Mass (kg)	Particulate Mass (kg)	Total Mass (kg)	Particulate Mass (kg)	Total Mass (kg)	Particulate Mass (kg)
Suspended Solids	3920	3920	2238	2238	666	666
Cu	1.33	1.04	1.16	.745	.904	.663
Hg	.026	.023	ND	ND	ND	ND
Pb	1.46	1.01	6.04	5.32	1.00	.502
Zn	3.20	2.20	2.74	1.76	.502	.050
Al	66.0	63.4	97.2	93.9	10.0	ND
Organic C	458	193	299	168	151	100
Total P	45.1	12.8	8.73	4.60	5.27	.502
Oils and Greases	330	NA	113	NA	65.3	NA
Chlorinated HC	ND	.460g	ND	.388g	ND	.150g

NA: Not applicable ND: Not determined

For all constituents except Pb and Al, the annual storm loading of particulates was less from the storm drain than from the combined sewer. However, with the nonstorm loading of the storm drain added to its storm inputs, the totals for Cu, organic C and chlorinated HC also exceeded those of the combined system, and the Zn loading was nearly as great. The parameter relationships were similar for the total loading (particulate + soluble), except that the C and P inputs were considerably more augmented for the combined sewer than for the storm drain due to the greater quantities of dissolved organics in the sanitary wastes of the combined system.

Receiving Water Monitoring

Turbidity Distributions of Effluent Plumes--

The distributions and movements of wastewater effluent plumes were monitored at CSO 023 and SD 7 during four different storms.

Background particulate distributions were also assessed at the control site during three storms. These measurements were made using a submersible light transmissometer system of the type originally described by Petzold and Austin (1968). This instrument provides a measurement of the volume attenuation coefficient, α, which is equivalent to the sum of the light scattering and absorption coefficients. The volume attenuation coefficient may in turn be related to common turbidity units using methods described by Austin (1973), who determined the relationship between α and Jackson Turbidity Units (JTU) to be approximately linear for natural water of moderate to good clarity.

For plume tracing, the light transmission (turbidity) data were reduced to sectional contour plots. These included aerial perspectives at various depths and onshore-offshore and longshore transects showing turbidity patterns from the perspective of a diver looking across the lake bottom.

Contrary to intentions, we were unable to obtain light transmission data for periods of lake stratification. The data presented here were collected during March and April of 1978 and February of 1979, and in all instances show the wastewater plumes to have been intersecting the surface. At the combined sewer outfall site this was due predominantly to the comparatively high wastewater temperatures; e.g., a difference of approximately 6°C was measured for a comparison of effluent and ambient water at CSO 023 on 4/4/78. For the storm drain the shallow discharge depth was the foremost determinant.

The various characteristics of the effluent plume dispersion observed at CSO 023 are summarized in Figures 5 and 6. Figure 5 shows the residual turbidity distributions 21.5 hr. from the time an overflow began on 3/7/78. The plume was seen as a lens of turbid surface water 150 m NE of the outfall, 2.7 m thick (Figure 6 (c)) and covering an area of approximately 11 acres. The extremely muddy water SW of the outfall (Figure 6 (a)) was temporarily uncontrolled drainage from a nearby building site. Interference from a large, nearshore storm drain was also observed, being particularly obvious during the storm of 2/6/79, for which contour plots of onshore-offshore transects clearly indicated a tongue of turbid wastewater moving offshore from the drain site 80 m SW of the CSO.

Considerable effluent interference was also observed in the turbidity distributions measured for storm discharges at SD 7. Substantial wastewater inputs from a 24-inch storm drain 345 m north of the SD 7 outfall were recorded for the storm of 2/6/79. The relevant areas of influence for the two outfalls are strikingly apparent in Figures 7 and 8, which give aerial turbidity distributions at one and four hours following a 10-hour storm discharge at SD 7.

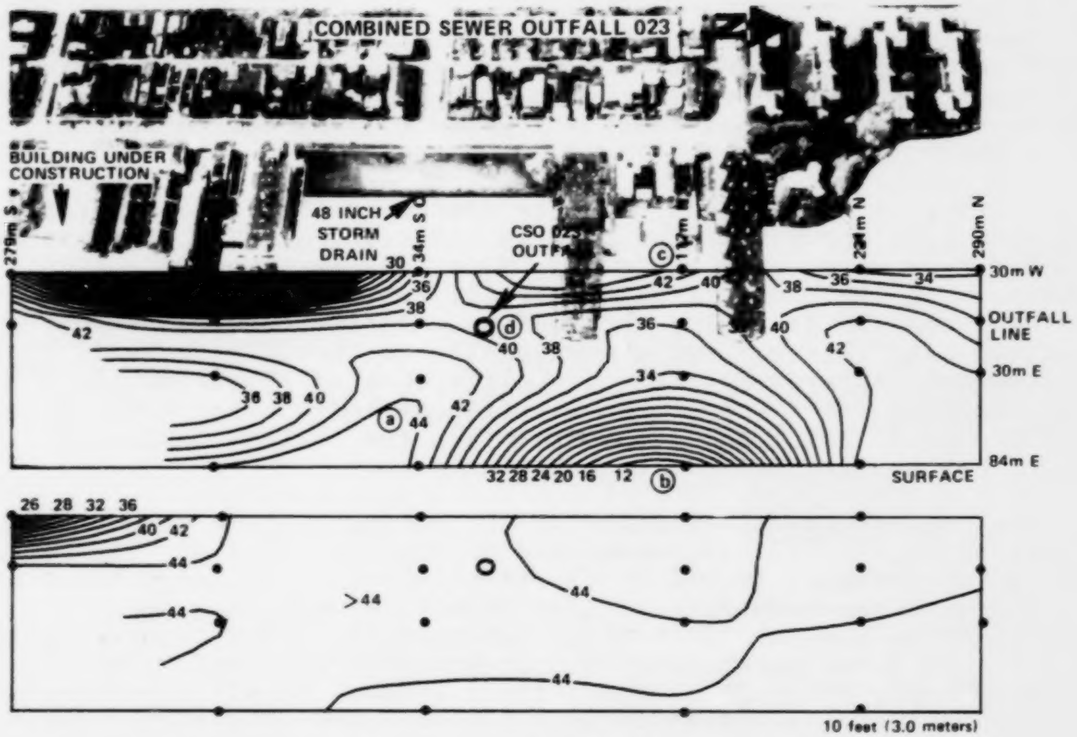


Figure 5. Aerial perspectives of contours of percent light transmission at Combined Sewer Outfall 023 - 1533-1645 hrs., 3/7/78. The lettered symbols indicate surface positions of the wastewater plume core at time of overflow initiation plus (a) 5 hr., 3/6/78; (b) 21.5 hr., 3/7/78 (distribution shown); (c) 3.5 hr., 3/24/78 - 3/25/78; and (d) 3hr., 4/4/78.

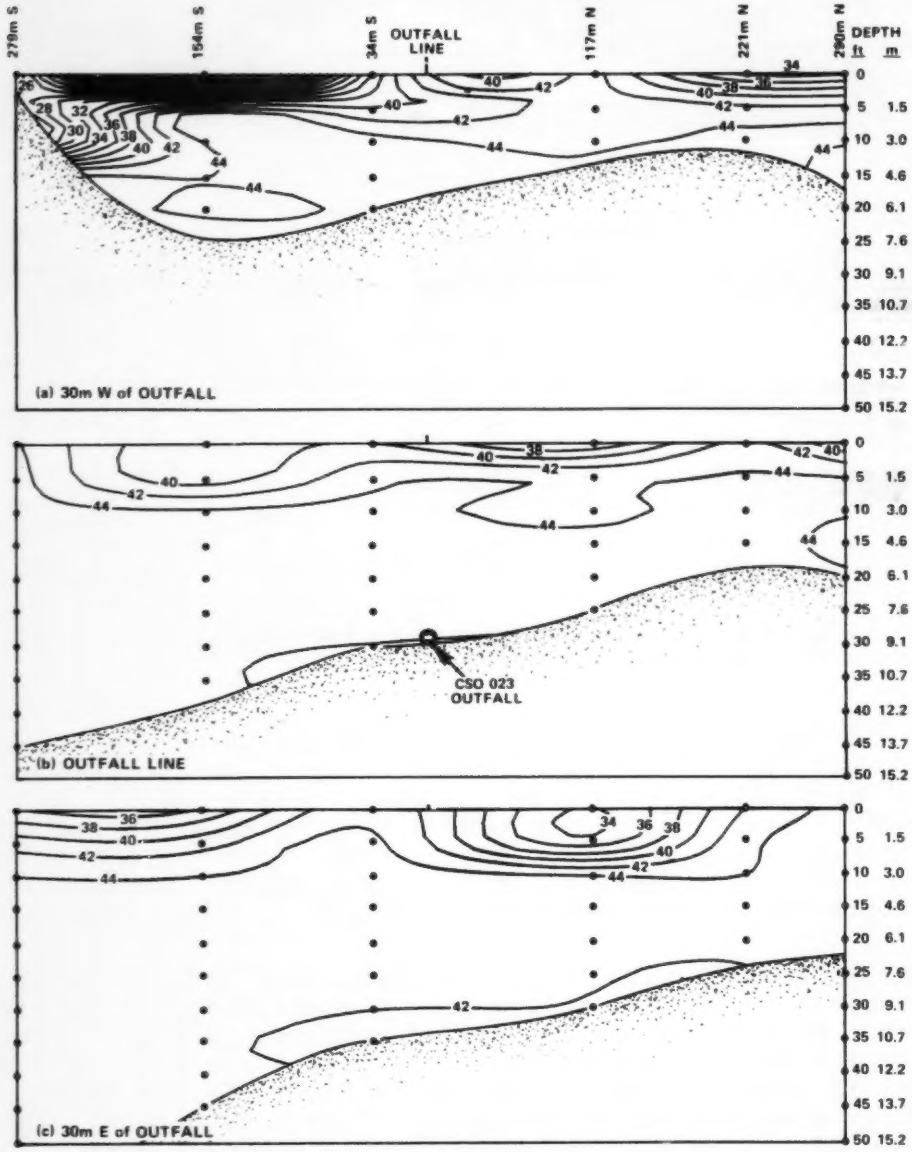


Figure 6. Longitudinal sections contours of percent light transmission at Combined Sewer Outfall 023 - 1533-1645 hrs., 3/7/78. The perspective is that of a diver facing shore.

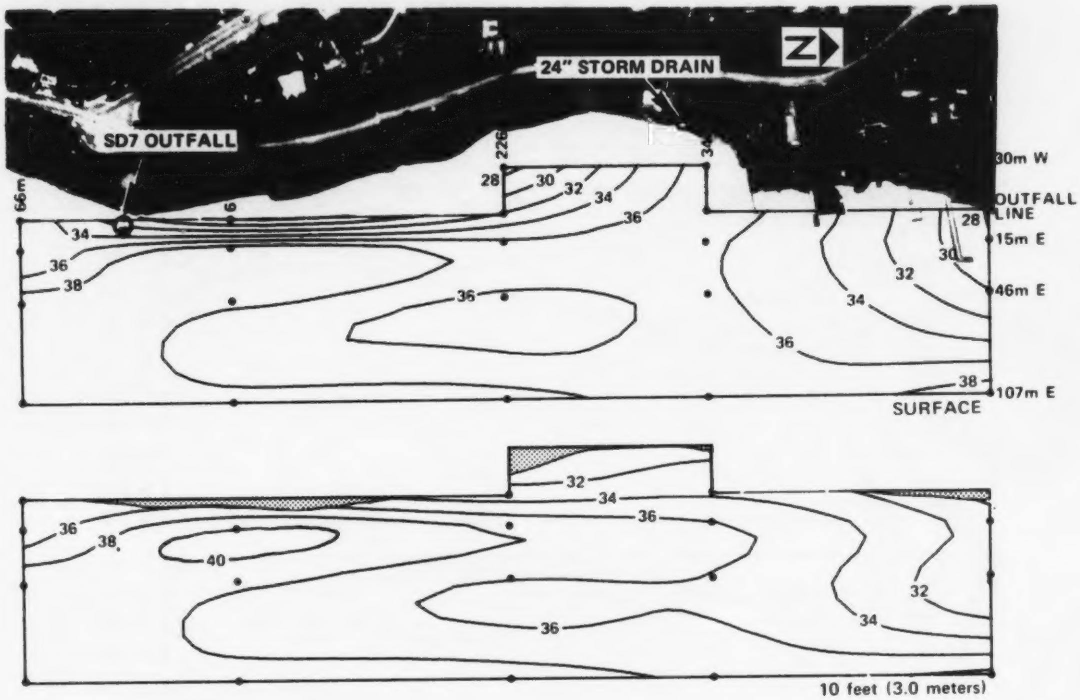


Figure 7. Aerial perspectives of contours of percent light transmission at Storm Drain 7 - 1915-2014 hrs., 2/6/78.

During the three-hour interim period the SD 7 plume began to dissipate and the effluent from the other outfall became more prevalent. The small cove between the two outfalls appears to have encouraged a local eddy in the nearshore circulation, resulting in a southerly movement of effluent offshore. There were also manifestations of this clockwise circulation pattern, and ov particulates discharged by the second storm drain, in storm data collected 3/6-3/7/78.

The surface position of an effluent lens as a function of time gives an incomplete indication of particulate circulation patterns. Much of the solid material present in these features apparently remains suspended for long periods, as evidenced by the lack of turbidity at mid-depths in Figure 6, and is ultimately advected out of the area. Figure 9, however, shows turbidity distributions for the onshore-offshore transect 61 m N ("downstream") of the SD 7 outfall, and indicates that a substantial local fallout and settling of particulates occur in an outfall area during an overflow; the settling pattern denoted by the figure shows a downslope movement of materials near the bottom typical of that observed near both the SD and the CSO. These observations correlate well with the measured distributions of sediment surface contaminants presented below.

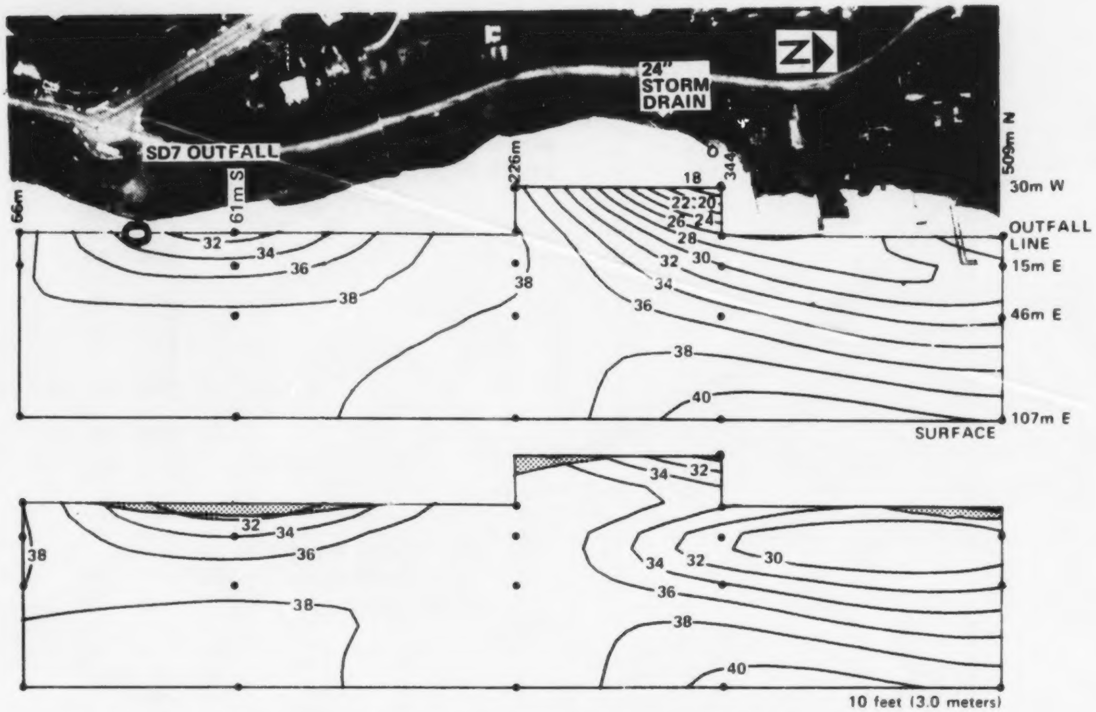


Figure 8. Aerial perspectives of contours of percent light transmission at Storm Drain 7 -

The control area (C 3) was also found to be inundated with

The control area (C 3) was also found to be inundated with suspended particulate matter. Light transmission contours drawn for storm data collected there indicate the presence of a deep, turbid layer believed to have been a longshore flow of settling particulates moving from the northeast. The most likely sources of this material were: 1) a series of 22 storm drains (including eight ≥ 24 in. diameter) carrying runoff from an area of active construction 0.7-3.8 km to the north, and 2) a creek which discharges highly turbid runoff from a 7,740-acre basin another 0.4 km beyond that (Figure 2).

The nature of turbid longshore advection through the C 3 area seemed to be quite variable - an extensive particulate layer observed entering from the north on 3/25/78 was on the lake surface, whereas that mentioned above was at depth. Regardless, the sediment quality analyses below implied only light settling of contaminated particles in this area, and the measured settling rate was substantially lower than for the CSO 023 and SD 7 areas.

Quantification of Settling Particulates--

Particulates captured by arrays of six sediment traps at each station were dried, weighed and analyzed for C, P, Pb, Cu, Zn and Al. The results are given in Table 4.

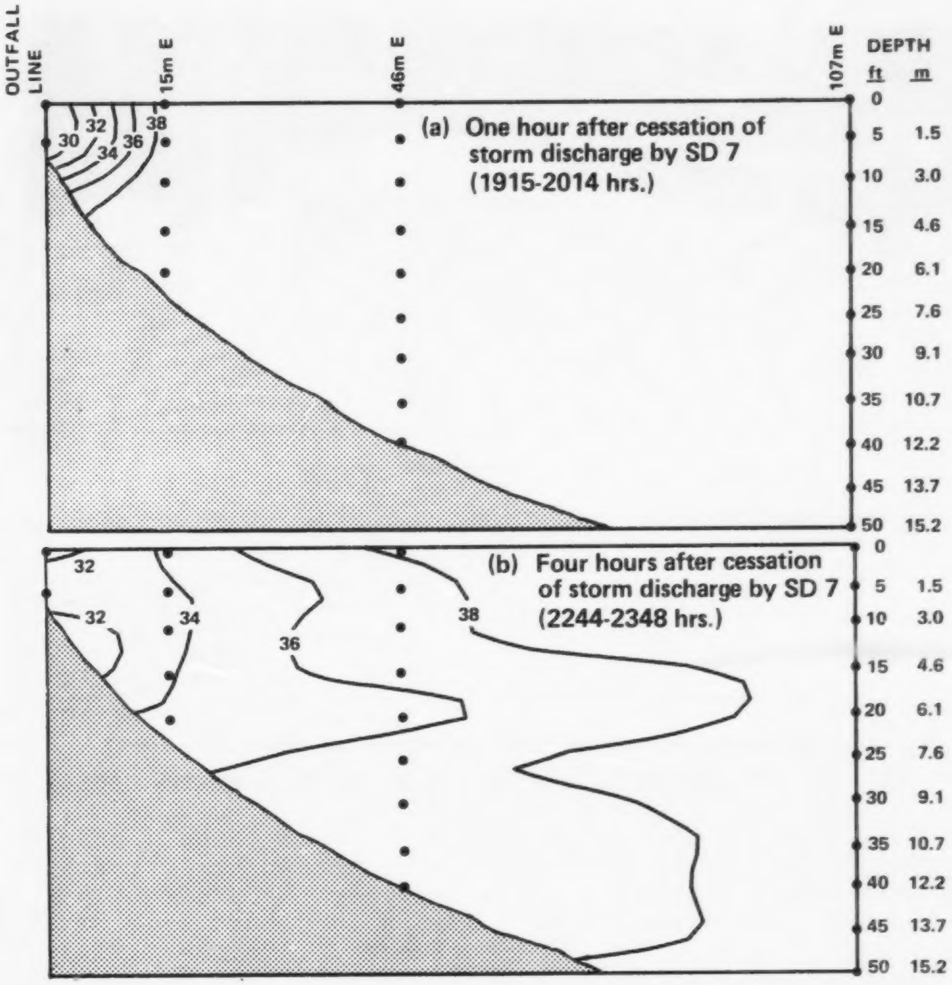


Figure 9. Transverse sections through a point 61m N of Storm Drain 7, showing contours of percent light transmission - 2/6/79. The perspective is that of a diver with the shore to his left.

The mean loading rates in g/m^2/day from 1/30/78 to 2/2/79 were .752 for CSO 023, .805 for SD 7 and .268 for C 3. A relative value for the adjacent profundal area is .661, which was determined by correcting a rate measured at 60 m depth (Birch, 1976) for the offshore movement of particulates entering the lake in the littoral zone. The offshore sedimentation rate was therefore less than nearshore values measured in the vicinity of the wastewater outfalls, but greater than that determined for the control area. A significant fraction of the suspended particulates in the study area was transient material which would eventually contribute to the sedimentation rate in the profundal zone.

TABLE 4. MEAN SEDIMENTATION RATES AND DRY WEIGHT CONCENTRATIONS OF SELECTED CONSTITUENTS ANALYZED IN SEDIMENT TRAP SOLIDS COLLECTED AT COMBINED SEWER OUTFALL 023, STORM DRAIN 7 AND CONTROL SITE 3, 1/30/78 - 2/2/79

	CSO 023		SD 7		C 3	
	$\bar{X}$	S_x	$\bar{X}$	S_x	$\bar{X}$	S_x
Sed. Rate (g/m^2/day)	.752	.714	.805	.696	.268	.199
Org. C (%)	9.58	3.85	9.29	4.53	9.34	6.08
Total P (%)	.252	.192	.212	.211	.273	.205
Pb (ppm)	366	104	521	355	245	97.1
Cu (ppm)	126	108	94.8	44.8	70.8	41.8
Zn (ppm)	349	152	286	127	244	125
Al (%)	4.26	1.07	4.70	1.13	3.72	1.09

The mean loading rates in g/m^2/day from 1/30/78 to 2/2/79 were .752 for CSO 023, .805 for SD 7 and .268 for C 3. A relative value for the adjacent profundal area is .661, which was determined by correcting a rate measured at 60 m depth (Birch, 1976) for the offshore movement of particulates entering the lake in the littoral zone. The offshore sedimentation rate was therefore less than nearshore values measured in the vicinity of the wastewater outfalls, but greater than that determined for the control area. A significant fraction of the suspended particulates in the study area was transient material which would eventually contribute to the sedimentation rate in the profundal zone.

The flux (or quantity settling through a given cross-sectional area per unit time) of particulate C, P, Pb, Cu, Zn or Al was computed as the product of the constituent concentration (wt./total wt. of solids) and the sedimentation rate. Compared to CSO 023 and SD 7, the control site had similar or lower concentrations of each settling contaminant and a significantly (at the 99% confidence level) lower sedimentation rate; the settling flux of all solid elements at C 3 was therefore significantly lower than at CSO 023 or SD 7.

The average concentrations of Pb in the sediment trap solids were significantly greater at SD 7 than at CSO 023--a relationship in keeping with the relative Pb discharge evaluations discussed previously (Tables 1 and 2). Cu and Zn had similar concentration hierarchies for settling particulates:

CSO 023>SD 7>C 3. The mean Al concentrations determined for solids settling near both outfalls were significantly higher than for solids from the control site, implying a greater input of erosional material.

Core Analyses of Sediment Contaminants--

To help further define trends of movement and accumulation of wastewater particulates in the receiving waters, sediment cores were collected at the transmissometer grid sites at Stations CSO 023, SD 7 and C 3. Fifty-six cores were dried, sectioned and analyzed for Pb, Zn, Cu and TOC. Their contour distributions in the 1-cm sediment surface layer are presented here in Figures 10 and 11.

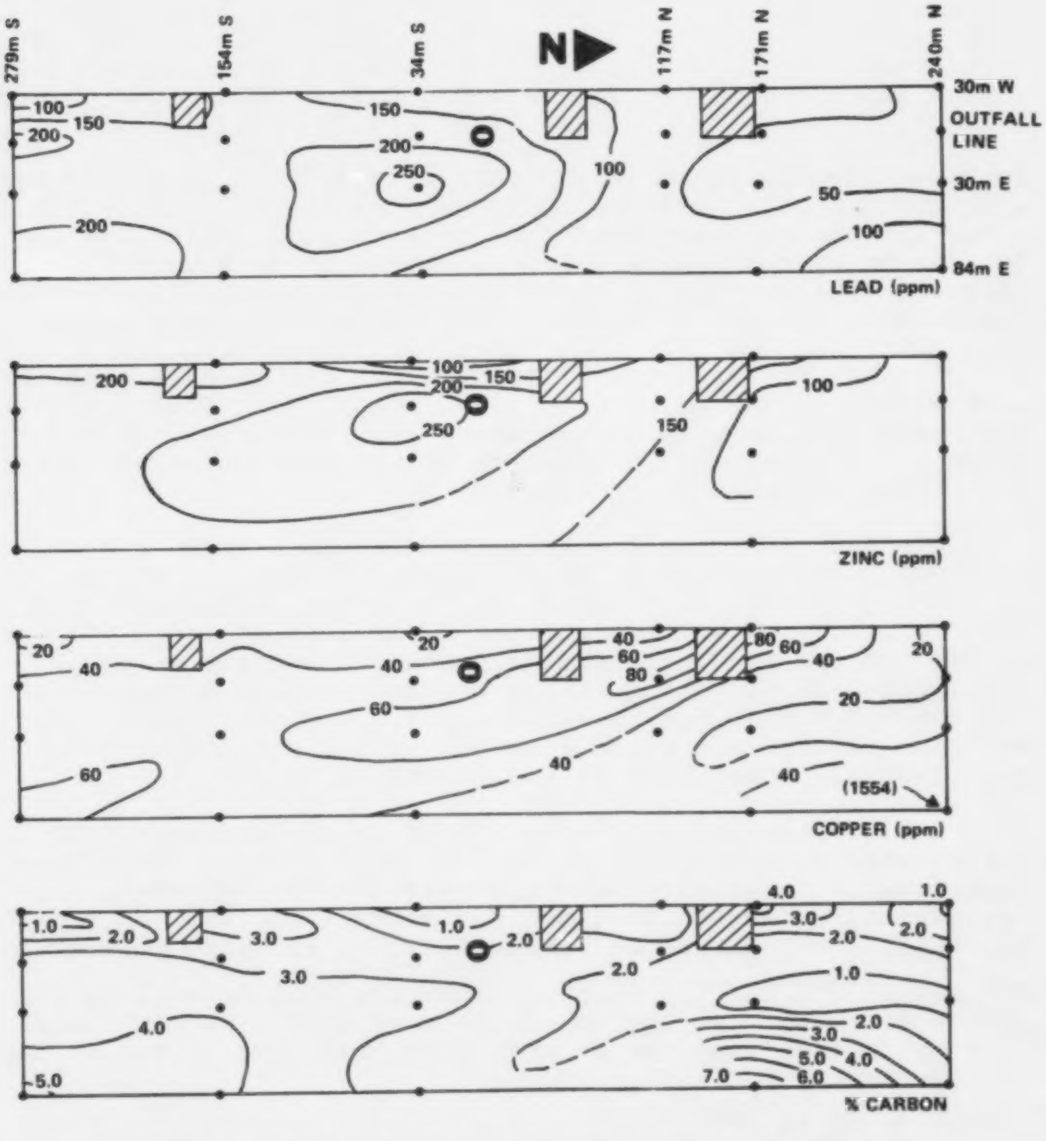


Figure 10. Dry weight distribution of lead, zinc, copper and carbon in the surface centimeter of sediments collected near Combined Sewer Outfall 023.

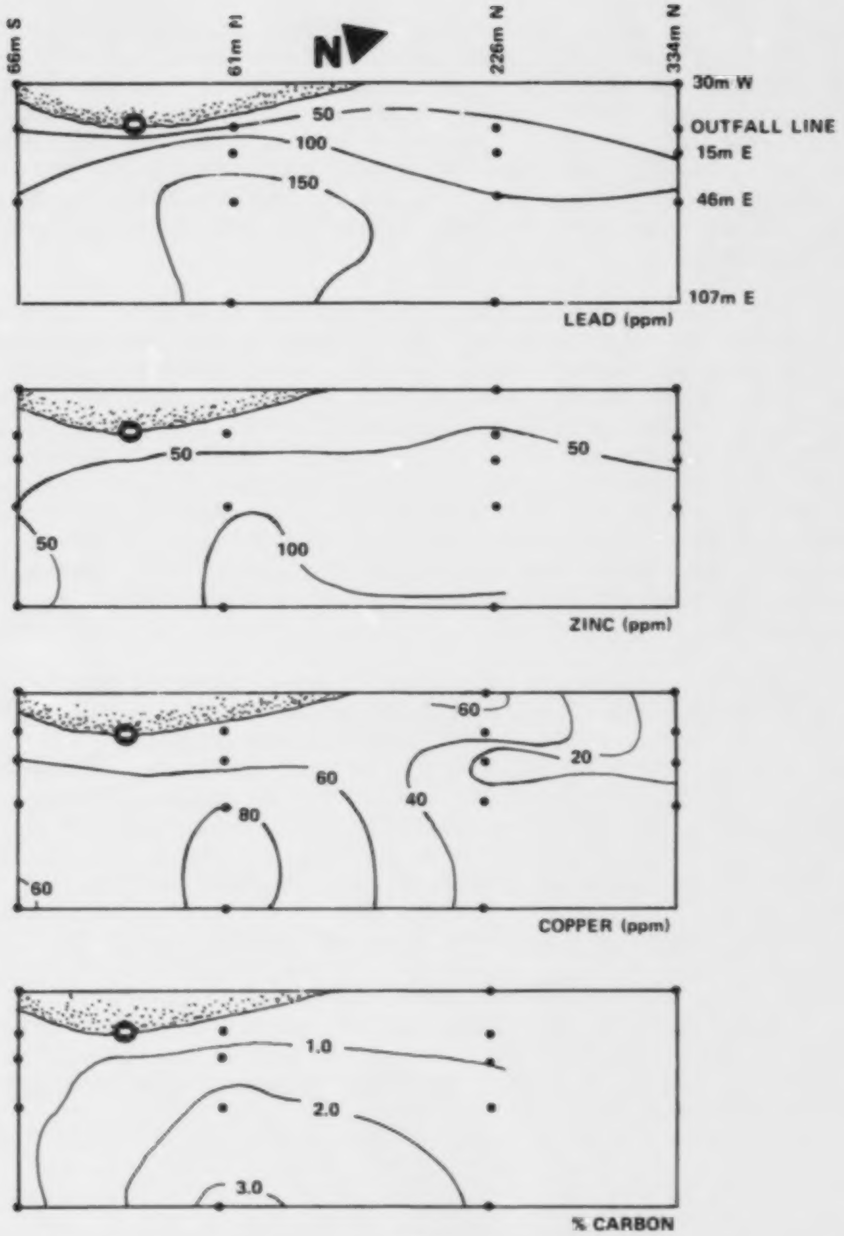


Figure 11. Dry weight distributions of lead, zinc, copper and carbon in the surface centemeter of sediments collected near Storm Drain 7.

The contours show enrichment of the sediments near the two outfalls, with apparent distribution modification from current action and near-bottom downslope streaming. The isopleths for Pb and Zn around CSO 023 show the effects of near-bottom turbidity plumes moving downslope to the southeast (refer to

Figure 4 for areal bathymetry). For Cu the concentration isopleths imply the additional influence of advective transport toward the northwest, where the particulates settled due to flow disruption by condominium support pilings. Although the prevalent direction of flow of subsurface currents near the CSO was not clearly defined by the light transmission studies, the northwesterly advection implied by the sediment distributions is logical as the most direct route to the flow outlet for Lake Washington.

The considerable differences in sediment distributions for Pb and Zn vs. Cu near CSO 023 support the hypothesis that the Cu-bearing effluent particulates were smaller and/or lighter, and thus more easily influenced by water motion. Carbon was found to be more widely dispersed than the metals, in keeping with the nature of the large, light carbon-bearing particulates. Correlation coefficients calculated for concentrations of Cu, Pb and Zn vs. C in CSO 023 sediments were 0.65, 0.42 and 0.24, respectively, implying that the relatively light Cu and C particulates have the most similar transport characteristics, and further confirming the Cu>Pb>Zn dispersion hierarchy suggested by Figure 10.

At SD 7, substantial advective losses of wastewater particulates from the sampling area result in a considerably different pattern of local deposition compared to that seen at CSO 023. Near the SD 7 outfall the sediment distributions of PB, Zn, Cu and C were all found to be quite similar (Figure 11), indicating the dominant influence of near-bottom downslope streaming just north of the outfall. Based on the strength of correlations between C and metal concentrations (Pb: r = .92, Zn: r = .89, Cu: r = .44) the dispersion hierarchy for SD 7 was determined to be Pb>Zn>Cu.

Circulation Analysis for Wastewater Solids--

Table 5 is a compilation of metals data for all types of particulates sampled at the three principal study sites. These numbers provide a useful aid for tracing the movements of wastewater solids.

TABLE 5. MEAN CONCENTRATIONS (1) AND CONCENTRATION RATIOS FOR SELECTED METALS IN PARTICULATES SAMPLED AT COMBINED SEWER OUTFALL 023, STORM DRAIN 7 AND CONTROL SITE 3

Location	Source	Mean Conc. (ppm) Pb	Zn	Cu	Pb:Cu	Zn:Cu	Zn:Pb
CSO 023	Effluent(2)	257	560	265	1.0	2.1	2.2
	Sed. Traps	366	349	126	2.9	2.8	0.9
	0-.5cm Sed.	129	168	39	3.3	4.3	1.3
	7-8cm Sed.	51	92	40	1.3	2.3	1.8
SD 7	Effluent(2)	2377	785	333	7.1	2.4	0.3
	Sed. Traps	521	286	95	5.5	3.0	0.5
	0-.5cm Sed.	89	61	47	1.9	1.3	0.7
	7-8cm Sed.	46	35	42	1.1	0.8	1.4
C 3	Sed. Traps	245	244	71	3.5	3.4	1.0
	0-.5cm Sed.	86	131	20	4.4	6.6	1.5
	7-8cm Sed.	21	51	7	2.9	7.1	2.5
Profundal(3)	0-1cm Sed.	192	192	46	4.2	4.2	1.0
	Pre-1900 Sed.	12	59	17	1.0	3.5	3.5

(1) Dry weight.
(2) Calculated from data in Table 1, as ppm of suspended solids in the effluent.
(3) From cores collected in the deep central basin of Lake Washington, between CSO 023 and SD 7 (Spyridakis and Barnes, 1976).

The concentrations of metals in the sediment trap collections were generally 2 to 5 times those of the sediment surface (0-0.5 cm) layer, implying selective removal to deeper areas, of the finer particulates with which the highest metal concentrations are typically associated (Guy and Chakrabarti, 1976). Sediment cores collected by Spyridakis and Barnes (1976) along an across-lake transect contiguous to CSO 023 and SD 7 clearly show an increase of Pb, Zn and Cu concentrations with lake depth. As is evident from a comparison of the CSO 023 and SD 7 sediment trap data with values for profundal (deep offshore) surface sediments (Table 5), however, the latter materials had been appreciably diluted by relatively uncontaminated secondary sources, including diatoms and river detritus. Spyridakis and Barnes have estimated that contemporary fluvial inputs (including wastewater discharges) are responsible for 26%, 90% and 53%, respectively, of the Pb, Cu and Zn entering the lake.

The selective transport tendencies of Cu, Pb and Zn at CSO 023 are also delineated by the concentration ratios given in Table 5. The Zn:Cu ratio increases as the particulates move from the outfall to the traps, and again, from the traps to the sediments; this is because of the higher losses of Cu through advective transport. The decrease of the Zn:Pb ratio represents an external input to this system - particulate Pb contributions from aeolian transport (Spyridakis and Barnes, 1976). The subsequent increase in the Zn:Pb ratio between the traps and the sediments implies a shorter local residence time for Pb. All of the sediments above 8 cm depth at CSO 023 have been deposited since the addition of tetraethyl lead to gasoline in the late 1920s, as is evident from their substantial Pb enrichment relative to pre-1900 (background) concentrations.

Estimates of median effluent particle diameter show that the wastewater particulates emitted by SD 7 are typically much smaller than those from CSO 023, facilitating advective losses; this helps to account for the comparatively precipitous drop of particulate metals concentrations between the SD 7 effluent and the sediment traps; the concentrations of all three metals in the effluent particulates were substantially higher at SD 7 than at CSO 023, but due to greater advective exports, sediment trap concentrations were higher only for Pb. The steeper bathymetry of the SD 7 sampling site also helps aggravate the offshore, near-bottom movement of effluent particulates.

Typical enrichment factors for total metals contributions from all sources to the nearshore surface sediments may also be derived from Table 5. For CSO 023, SD 7 and C 3, respectively, they are: Pb - 11 times, 7 times, 7 times; Zn - 3 times, none, 2 times; and Cu - 3 times, 3 times, none. Overall, the SD 7 area was not found to be much different than the control site in this sense, a conclusion statistically confirmed by an analysis of variance for mean station concentrations of Pb, Zn, Cu and C in the 0.5 cm sediment surface layer. This is due predominantly to the rapid movement of effluent particulates out of the SD sampling area and to the contamination of the control site by longshore advection.

SUMMARY

The analyses of sediments collected near ten combined sewer outfalls (CSOs) and ten storm drains (SDs) discharging into Lake Washington revealed their widespread enrichment with heavy metals, chlorinated hydrocarbons and organic wastes. This was particularly true of the western nearshore region, where all of the CSOs were located. Intensive comparative studies were carried out at one representative CSO and one SD having similar drainage basins and rainfall. Pollutant loading estimates indicated that, whereas mean storm concentrations for most parameters were greater for the CSO, the total annual loading was greater in most instances for the SD, due to its continuous and therefore greater volume of discharge. Effluent turbidity patterns and metals concentration ratios for effluent solids,

settling particulates and sediments demonstrated significant local deposition of wastewater particles. The effects of near-bottom,offshore transport and longshore advection were also indicated, with different dispersion patterns seen at the SD than at the CSO.

REFERENCES

APHA. 1975. Standard Methods for the Examination of Water and Wastewater. 14th Edition. Amer. Pub. Health Assoc., Amer. Water Works Assoc., Water Poll. Cont. Fed., Wash., D.C. 1193 pp.

Austin, R.W. 1973. Problems in measuring turbidity as a water quality parameter. Paper presented at Environmental Protection Agency Seminar on Methodology for Monitoring the Marine Environment. October 16-18, 1973. Seattle, Wash.

Birch, P.B. 1976. The Relationship of Sedimentation and Nutrient Cycling to the Trophic Status of Four Lakes in the Lake Washington Drainage Basin. Ph.D. dissertation, Univ. of Washington, Seattle.

Bortleson, G.C. and G.F. Lee. 1972. Recent sedimentary history of Lake Mendota. Wis. Env. Sci. and Tech. 6:799-808.

Brenner, R.M., R.J. Morrice and R.G. Swartz. 1978. Effects of Stormwater Runoff on the Juanita Creek Drainage Basin. Report for the Municipality of Metropolitan Seattle. 59 pp.

CH2M/Hill. 1974. Storm Water Outfall Study of Lake Washington. Report for the City of Seattle. 50 pp.

Guy, R.D. and Chakrabarti. 1976. Studies of metal-organic interactions in model systems pertaining to natural waters. Can. J. Chem. 54:2600-2611.

Petzold, T.J. and R.W. Austin. 1968. An Underwater Transmissometer for Ocean Survey Work. SIO Ref. 68-9, Scripps Inst. of Oceanography, Visibility Laboratory. Univ. of Calif., San Diego.

Phillips, E.L. 1968. Washington Climate. Coop. Ext. Serv., College of Agriculture, Wash. State Univ., Pullman. 65 pp.

RIBCO. 1974. Environmental Management for the Metropolitan Area, Part I. Water Resources Appendices. Compiled by CH2M/Hill for the River Basin Coordinating Comm., Seattle.

Spyridakis, D.E. and R.S. Barnes, 1976. The Effects of Waste Water Diversion on Heavy Metal Levels in the Sediments of a Large Urban Lake. Compl. Rpt. to OWRT (Proj. No. A-070-WASH). Dept. of Civil Engineering, Univ. of Wash., Seattle.

THE ECOLOGICAL EFFECTS OF URBAN RUNOFF ON STREAM COMMUNITIES

Donald B. Porcella
Tetra Tech, Inc.
Lafayette, California

ABSTRACT

A literature review of the ecological effects of urban runoff on streams indicates a dearth of principles for evaluating the impact of urban runoff or any pollutant on streams and a serious lack of studies that determine urban runoff impacts on communities of stream ecosystems. The uniqueness of urban runoff impacts results from the nature of pollutant input to streams --large magnitued variation/stochastic occurrence. Most stormwater research has been directed towards determining runoff quantity and quality including the fates of specific pollutants such as heavy metals, nutrients, salts, toxic substances, and bacteria. Demonstration of the impacts of these materials on stream communities is necessary to justify costs of managing stormwater effects.

Several approaches for analyzing ecosystems are suggested, such as stream community analysis and biogeochemical cycling of elements (carbon, nitrogen, phosphorus). Three published case studies of stream ecological impacts typify the effects of urban runoff inputs resulting in a hypothesis that large scale variations and instabilities of stream pollutant input and concentration would result in greater impact to stream communities than steady inputs. This concept is discussed and applied to urban runoff analysis and management.

INTRODUCTION

Materials in urban runoff impact stream ecosystems in several specific and unique ways that distinguish those impacts from other linked systems. Published literature was used in an attempt to determine whether those unique features result in ecological differences that can be distinguished by appropriate ecosystem variables.

THE URBAN NPS/STREAM SYSTEM

By definition, materials in urban runoff originate from nonpoint sources (NPS). These NPS materials are transported by water generated as a stochastic (random, but characterized by time of the year) hydrologic event, are affected significantly by temporal and spatial factors that have preceded the hydrologic event, and impact aquatic ecosystems as discrete, largely receiving-system-independent input. Urban runoff impacts are similar to other aquatic ecosystem impacts with respect to the relationships between activities and materials produced, the materials themselves, and interactions within the aquatic ecosystem that are not a function of the input. Control of urban runoff occurs in the watershed but ecosystem impacts are the basis for assessing the need for such controls.

The system watershed can be visualized as consisting of separate but somewhat interrelated phases or processes: 1) the production of waterborne materials in the watershed which can enter the aquatic ecosystem, 2) the hydrologic event, 3) the transport system, and 4) the aqueous environment and its community (Figure 1). Urban runoff impacts on the aquatic ecosystem are characterized by large magnitude, random inputs of materials that affect the ecosystem. Thus, they are similar to "spill" events and other "catastrophic" types of events.

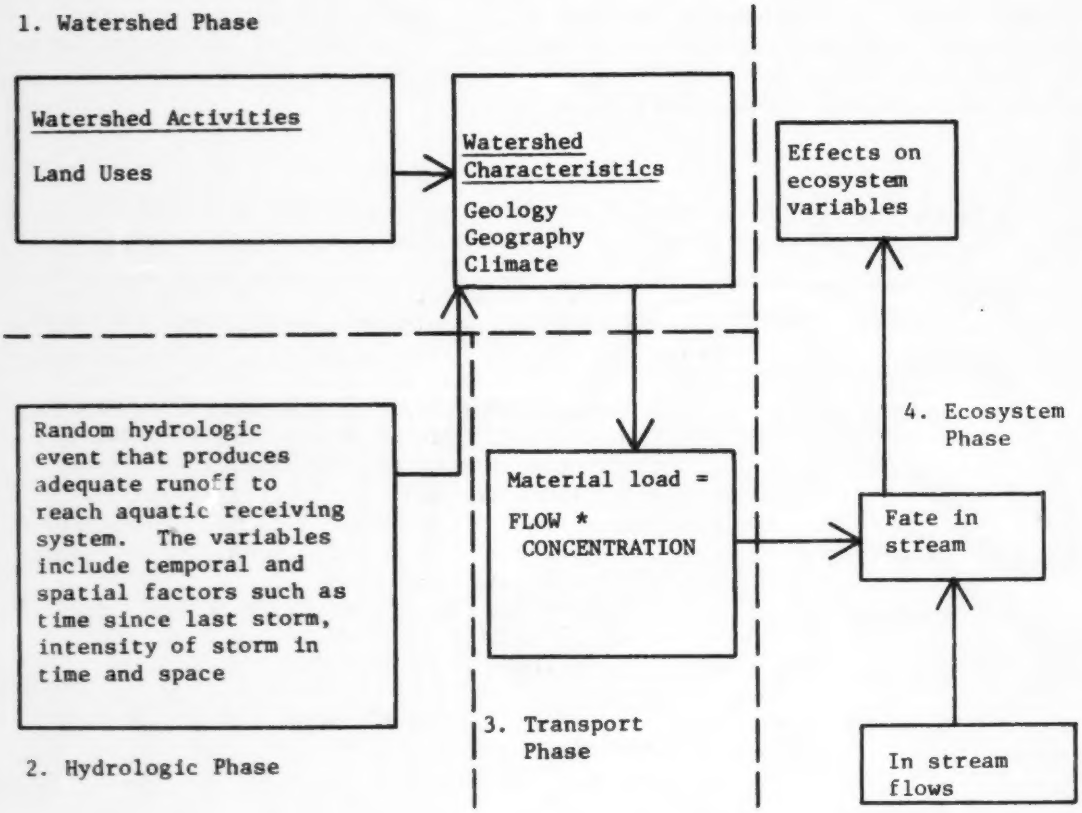
Most studies of urban runoff are concerned with the first three phases of the stream ecosystem and with water quality in the receiving environment. In this paper it is postulated that the urban runoff impact on the stream community is different from other types of inputs and that, therefore, the community should be evaluated differently. Although methods of study would be similar, interpretation of timing effects is different. The need to regulate or treat urban runoff depends on accurate assessment of impacts on the aquatic community. Without this assessment, costly and complex NPS controls may not be justified. Only stream ecosystems are discussed in this paper. Lakes generally serve as at least partial sinks for input materials. Lakes may not differentiate between urban NPS loadings and continuous loadings such as wastewaters, while streams are impacted and then have an opportunity to recover as they are flushed by upstream water.

CASE HISTORIES OF RANDOM EVENT AND MAGNITUDE IMPULSE IMPACTS ON STREAMS

The selection of case studies was not intended to be exhaustive nor restricted to urban runoff events. Generally, the approach used was to locate stream impact analyses and use those which illustrate the following assumptions about urban runoff:

- Urban runoff often alters stream flow patterns.
- Pollutant inputs exert impacts related to accumulation by the biota which in turn is based on pollutant concentration.
- Pollutant concentrations vary extensively and relatively rapidly because of the mode and timing of runoff input.

Justification for this approach is the lack of published studies relating urban runoff impacts to stream communities and the very few related to impulse inputs to stream ecosystems. Approximately 1,200 reports and papers on urban runoff have been critically evaluated (Porcella and Sorensen, 1979).



1. Watershed Phase
Watershed Activities
Land Uses
Watershed Characteristics
Geology
Geography
Climate
Effects on ecosystem variables
Random hydrologic event that produces adequate runoff to reach aquatic receiving system. The variables include temporal and spatial factors such as time since last storm, intensity of storm in time and space
Material load = FLOW * CONCENTRATION
4. Ecosystem Phase
Fate in stream
In stream flows
3. Transport Phase
2. Hydrologic Phase

Figure 1. The hierarchial stream/watershed system that produces material flow to aquatic ecosystems.

It was estimated that 2% related to ecological impacts, 40% concerned modeling, hydrology and hydraulics, 35% concerned urban runoff measurements and 23% applied to water quality measurements in the stream.

Although BOD/DO models have been used to assess urban runoff and other NPS impacts on streams (Rickert et al., 1975), these models do not reflect response of higher organisms. Until stream ecological models can be developed to assess nonpoint source impacts on the stream community, simpler methods comparing upstream/downstream, before/after, and between similar environment community variables must be used. As a part of the review, useful target variables were identified that would reflect stream biotic response to pollutant inputs (Table 1).

Table 1. Variables identified with properties that describe the state or condition of ecosystems

Properties	Variables* (measured per standard unit area)
I. Environmental	Volume, pressure, abiotic mass flow rates, system attributes (homeostasis-feedback, storage, loss), physical attributes (topography, geology, . . .), habitat.
II. Structure	
Biomass	Diversity, number, mass, chlorophyll-a
Energy	Calories
Materials	C, N, P, ..., total mass
III. Function	
Biomass	Change in diversity, number, weight, competition, resilience, resistance
Energy	Energy transfer, efficiency, P/R ratios
Materials	Elemental turnover times, P/R ratios, limiting nutrient

*Can be applied at any or all levels: community, trophic, population, organism, dominance, keystone, guild (functional groups) or habitat.

FLOODING EFFECTS ON FISH COMMUNITIES

Harrell (1978) analyzed effects of flooding on fish diversity for Devil's River in Texas. A fortuitous flood event (ninth largest on record) occurred midway during Harrell's study to assess the fish community of the river. Marked changes in habitat occurred as a result of the flooding; distinctness of spring, riffle, pool, channel and intermediate habitats generally were blurred and riffle-like habitat-types were more frequent. Fish community diversity declined. Harrell suggests that the ecological plasticity of the communities allowed shifts in the roles and associations of fish species and thus allowed the community to be maintained. In this respect, the community is stable; naturally stressful environments (Slobodkin and Sanders, 1969; Orians, 1975) would result in dominance and maintenance of a few species.

EFFECTS OF TOXICANT SPILLS

Although there are many spill impact studies on marine ecosystems, few similar types of studies have been done in freshwater systems. Spill impacts are random impulse type events and therefore are analogous to urban NPS run-off impacts on streams, although they occur less frequently.

Cairns and Dickson (1977) describe the Clinch River (Virginia) as impacted in this manner. On June 10, 1967, caustic wastewater (pH = 12) from a fly ash pond associated with a power plant adjacent to the Clinch River were released through a collapsed dike and caused extensive damage to the aquatic community. The pond inflow was 40 percent of the normal Clinch River flow at that time (Cairns *et al.*, 1970). In 1971 an acid spill occurred. Meanwhile during this period, day-to-day operations of the power plant and periodic flooding had short term effects on the aquatic biota. The authors and co-workers assessed benthic macroinvertebrates at 21 ecologically similar stations during the years 1969, 1970, and 1971. The data analysis was based on numbers, biomass, diversity, and presence/absence (cluster analysis).

Although approximately 200,000 fish were killed by the spill, Cairns and co-workers focused on benthic invertebrates because they

1) are relatively sessile and cannot avoid stress;

2) have long and complex life cycles;

3) are important members of the food web and affect related organisms;

4) are sampled by techniques that are more reliable than for other organisms;

5) yield more biological information obtained per dollar invested.

The initial measurements (June 20, 1967) were made by State of Virginia employees and Cairns and associates analyzed these data to show that the number of taxa and organisms per unit area were severely impacted by the spill (Table 2).

The authors noted that the benthic community was 8 percent midges at Station 1 but increased to more than 60 percent at damaged stations, illustrating tolerance to stress. By Station 10, the midges were less than 30 percent. The pattern at these lower stations was that of lesser impact (due to dilution and physico-chemical reactions).

Recovery was not rapid and was still ongoing for the July-December, 1969 sampling (Table 3). The biotic density was considerably depressed but diversity had reached control levels. Unfortunately no data on diversity were presented for the 1967 data so it was not possible to analyze diversity as a function of time (e.g., Cornell et al., 1976). The longer life cycle and non-motile organisms had not returned by 1969; thus recovery is a function of recolonization time which is controlled by life cycle and motility. Recovery was essentially complete by 1971.

STREAM COMPARISONS - MICHIGAN

Ball et al. (1973) studied three streams in Michigan to show pollutant impacts on stream communities: 1) the Jordan River was least developed with the downstream sitd differing from the upstream site because of a fish hatchery effluent; 2) the Au Sable River was affected chiefly by recreational use but also by a previously thriving lumber industry and forest fires; 3) the Red Cedar River was in a more urban/industrial region and, although having received waste effluents directly in past years, urban runoff and some industrial inputs were more common. Generally water quality at upstream stations was better than downstream.

The authors and their colleagues had performed a variety of measurements to compare with material inputs to the rivers. Although they concluded that the rivers were geochemically similar, the slope of the Red Cedar River was steeper. Generally, total organic carbon, non-carbonate hardness and total phosphorus were greater at the lower station of the Red Cedar River than the other stations. Suspended matter was considerably greater for both stations on the Red Cedar River than for the other rivers. Generally toxicants (metals, pesticides and PCB's), inorganic nitrogen compounds and chloride were greater on the Red Cedar. Although data on macrophytes and fish were discussed, those data were not directly comparable to other measures because the same species were not always present at all the stations.

The most important variables of comparison are listed in Tables 4 and 5. The Au Sable River seemed most productive based on diel DO fluctuations although the gross productivity of the Red Cedar River was greater. A comparison of average P/R ratios indicated that the better quality rivers (Jordan and Au Sable) had fairly typical values for clean midwestern rivers

Table 2. Ash pond spill (June 10, 1967) effects on Clinch River (Virginia) benthic invertebrates (adapted from Cairns et al., 1970)

Station No.*	Description	RM	Taxa	Total Benthos/m^2
1	Upstream (reference)	+21.0	24	5,600
2	Upstream but backwater effects	+ 0.5	9	320
3	Downstream	- 0.3	0	0
4	Downstream	3.5	5	2,900
5	Downstream	9.5	11	3,060
6	Downstream	12.5	15	7,000
7	Downstream	17.2	8	6,900
8	Downstream	30.5	11	1,900
9	Downstream	40.2	22	8,100
10	Downstream	56.5	18	860
11	Downstream	65.5	22	2,800
12	Downstream	77.5	32	11,500

*Sampled June 20, 1967; RM = river mile from spill.

Table 3. Recovery of Clinch River benthic invertebrates of the ash pond spill, Virginia (adapted from Cairns _et al_., 1970)

Station No.*	Description	RM	Taxa	Total Benthos/m^2	Diversity (H", log 2)
1-4	Above the spill (n=4)	+ 1.5 to +26.5	48-54	260-1600	2.97-4.03
7		- 0.2	43	510	2.92-4.21
8		- 0.9	33	26	3.64-4.35
9		- 3.9	36	43	2.88-4.01
10		- 9.5	42	86	3.19-4.06
11		-18.5	46	200	3.61-4.17

*Sampled July-December, 1969; RM = river mile from spill.

Table 4. Comparison of stream quality variables for different rivers in Michigan (adapted from Ball *et al.*, 1973).

Variables	Upper Jordan	Lower Jordan	Upper Au Sable	Lower Au Sable	Upper Red Cedar	Lower Red Cedar
DO difference, mg/ℓ (maximum observed)	2.5	2.5	4.5	9.0	5.5	5.2
DO maximum, mg/ℓ	9.5	13.0	12.0	13.0	10.5	9.0
Mean Flow, m^3/sec	1.8	5.5	1.7	2.2	1.9	5.4
CV, percent	13.0	16.0	27.0	27.0	53.0	110.0
Diatoms (artificial substrates)						
Maximum Observed Production, mg C/cm^2·day	120.0	230.0	70.0	80.0	100.0	440.0
Genus Diversity (H", log 2)	2.10	2.15	1.78	1.35	1.71	1.92
Invertebrates						
Benthic Species Diversity	4.33	3.37	-	2.76	-	1.70
CV, percent	6.0	18.0		31.0		27.0
Drift, gC/m (width)·day	8.0	7.0	17.0	12.0	10.0	9.0
Average P/R Ratios	0.9	0.8	0.9	0.9	1.3	0.6

CV = coefficient of variation.

Table 5. An energy budget for the three rivers (adapted from Ball et al., 1973).

Energy Source and Units (g C/m^2 · d)	Upper Jordan	Lower Jordan	Upper Au Sable	Lower Au Sable	Upper Red Cedar	Lower Red Cedar
Periphyton Production	0.03	0.08	0.03	0.06	0.06	0.08
Macrophyte Production	0.03	0.01	-	0.29	-	0.01
Sum or Periphyton and Macrophyte	0.06	0.09	0.03*	0.35	0.06*	0.09
Gross Photosynthesis	0.60	1.20	1.20	2.50	4.00	2.80
Community Respiration	0.70	1.60	1.30	2.90	3.10	5.00
Terrestrial Input**	323	762	267	268	797	236
Fine Particulate Organic Matter**	3,900	13,200	10,700	10,900	15,900	39,600
Dissolved Organic Matter**	18,200	22,800	32,300	22,200	65,800	2.6,400

* Value is low because macrophyte production was not estimated.
** Lineal inputs, gC/m · day.

while the Red Cedar River was the most variable and had values outside of the normal range of P/R ratios. This may have been associated with the considerable input of organic matter in the lower Red Cedar, possibly from urban runoff.

Community structure as measured by diversity varied considerably. Although diatom diversity was varied considerably. Although diatom diversity was greater for the lower Red Cedar River than the Au Sable, and for invertebrates the lower Au Sable was greater than the lower Red Cedar, both stations on the Jordan were considerably greater than those of both the other rivers. Also, the error associated with the estimated diversity for the Au Sable and the Red Cedar Rivers was greater than for the Jordan. This observation relates closely to the theoretical development of diversity change as proposed by Cornell et al. (1976). Based on these data (Tables 4 and 5) one could conclude that all of the variables, DO, diversity (periphyton, invertebrates), and productivity could be used to assess urban NPS runoff impacts in freshwater streams.

NPA RUNOFF IMPACTS ON STREAM COMMUNITIES

Chisholm and Downs (1978) studied the benthic invertebrate community of a small West Virginia stream which received sediment from construction of a super-highway. In comparison with a nearby control stream, the diversity index, generic count, and total count of invertebrates indicated severe reduction or destruction of the benthos in the impacted stream. The greatest degradation occurred in areas of highest sediment movement. When construction ceased, the benthic population of the impacted stream recovered to a comparable level with the control stream. Colonization was from upstream, unimpacted tributaries.

Except in streams subject to combined sewer and sanitary overflow discharges, Ragan and Dieteman (1975) found streams in areas of Maryland, urbanized during and after the 1950's, to be of good quality. This was not expected but the authors attributed this to the lack of sanitary discharges in the area, and to the large (>10 square miles) size of the drainage basins being studied. Even though traditional water quality parameters did not show dramatic changes, pollution sensitive fish species (e.g., Rosyside Dace) were no longer found in urbanized areas. A nearby unurbanized stream had the same 21 species of fish that were present in 1912.

DiGiano et al. (1975, 1976) directed their study in Greenfield, Massachusetts, toward both short-term impacts on water quality and the longer term disruptions of the benthic macroinvertebrate population caused by urban runoff. Using weekly grab samples from the Green River, they found that total P, turbidity, chloride, and total coliform concentrations increased with runoff events. They were unable to show conclusively that BOD, TOC, and oil and grease concentrations increased with runoff. It was suggested that more detailed analyses of both discharge and concentration be used in determining the mass loading for relatively short (3.2 mile) river reaches. In an analysis of three specific storm events sampled at 15 or 30 minute intervals, they found total pollutant loading variations of 50 to 100 times from storm to storm. Distinguishing between combined sewer overflows and stormwater runoff events is also important, particularly for bacteriological studies.

The benthic diversity (Brillouin's) of the stream macroinvertebrate community remained constant upstream. However, downstream in the urbanized areas of the stream, reduced diversity and a community shift to more pollution tolerant (polysaprobic) species occurred. In addition, the seasonal variation in diversity increased at downstream stations. The physical character of the stream bottom and availability of substrate deteriorated due to sedimentation. Sedimentation did not explain all of the changes in benthos. Evaluating the effect of contaminants from natural and man-made sources, bioaccumulation of metals through the detritus-based food chain, and stress from these heavy metals on benthic invertebrates required further investigation to determine cause and effect relationships.

CASE HISTORY SUMMARY

The major conclusions from these case histories are:

- macroinvertebrates and periphyton seem to be effective for monitoring impulse impacts on streams;
- diversity and biomass are often used for the analysis of monitoring data but they are not always adequately sensitive;
- water quality variables do not always reflect stream impacts directly. The biota must be assessed by measuring the quantitative effects of impulse inputs and contaminants.

A scenario based on the cited case histories and other knowledge of the system illustrates the variables (hydrologic, physical, chemical, biological, ecological) for determining urban NPS runoff impacts on stream ecosystems. A storm occurs at a specific time for a specific interval and intensity. Water begins to run off depending on antecedent storms, soil moisture and impervious surfaces. The pickup of contaminants varies with antecedent "washings", velocity of runoff and natural and cultural activities in the runoff basin. Water containing contaminant materials enters the stream diffusely, or at a point or several points, and over an interval and pattern controlled by the transport system and the storm event. This pattern provides the random event/magnitude impulse of contaminants.

The impulse of contaminants is diluted within the stream and has variable impact depending on flow, peak concentration of contaminants, duration of exposure and life cycle period of the organisms making up the community. Survival of specific organisms/taxa/niche is controlled by such events.

Eliminated community components can be replaced by recolonization but the interval depends on out-of-stream or upstream sources and physiological variables (length of reproductive cycle, motility). Continual contaminant inputs may prevent recolonization. Where storms are more frequent, inputs approach the limiting condition of point sources such as wastewater discharges from domestic and industrial sources.

Other niches may be opened by short-term infrequent inputs but generally communities become simpler, nutrient cycling speeds up (shorter turnover times), and ecological efficiency decreases. The variables that seem most useful in reflecting these processes are not necessarily biomass and diversity but include those processes associated with community functions; energy flow and material cycling should also be measured to determine their value as an assessment tool.

A PROBABILITY HYPOTHESIS FOR ECOLOGICAL RESPONSE VARIABLES

Throughout this paper, it has been asserted that random event and magnitude impulses to stream ecosystems would result in changes in ecological variables related to community structure and function. In this section, an approach for assessing such impacts is proposed. Because the spatial distribution (Slocomb and Dickson, 1978) and the probability distribution of runoff events can be determined, a stochastic approach is suggested. Some application of stochastic models to water quality data illustrate the feasibility of this approach.

Krishnan *et al.* (1974), Padgett *et al.* (1977), and Pamanabhan and Delleur (1978) applied this approach to BOD modeling in streams and thus were able to calculate the mean DO in a stream, the error about the mean, and the probability of violating a specific criterion or standard. In all three reports, data generated from storm runoff models were used because actual data were insufficient. Temperature modeling has been used to assess the effects of catastrophic events such as climatic and flow variations which result in extreme water temperatures. The model provided a reasonable duplication of actual data (Morse, 1978). It is simple to see how such events could result in large-scale community changes but it is difficult to assess the magnitude of the ecological response and even more difficult at this time to develop the relationships between the impulse and the response. For example, Slatkin (1978) used a model study to show that community stability existed only when the time scale for environmental change is roughly comparable to the average response time due to those changes. Means and variances did not determine extinction tendencies.

Material (nutrient, toxicant) and energy (allochthonous reduced carbon, light, temperature) inputs to streams as impulse loadings vary seasonally. Such impulses are exemplified for a hypothetical seasonal cycle for a fish in Figure 2. Two sensitive stages are illustrated: 1) egg growth and larval development requires dissolved oxygen (the DO is utilized in degrading organic matter in urban NPS runoff); 2) fry development requires lower

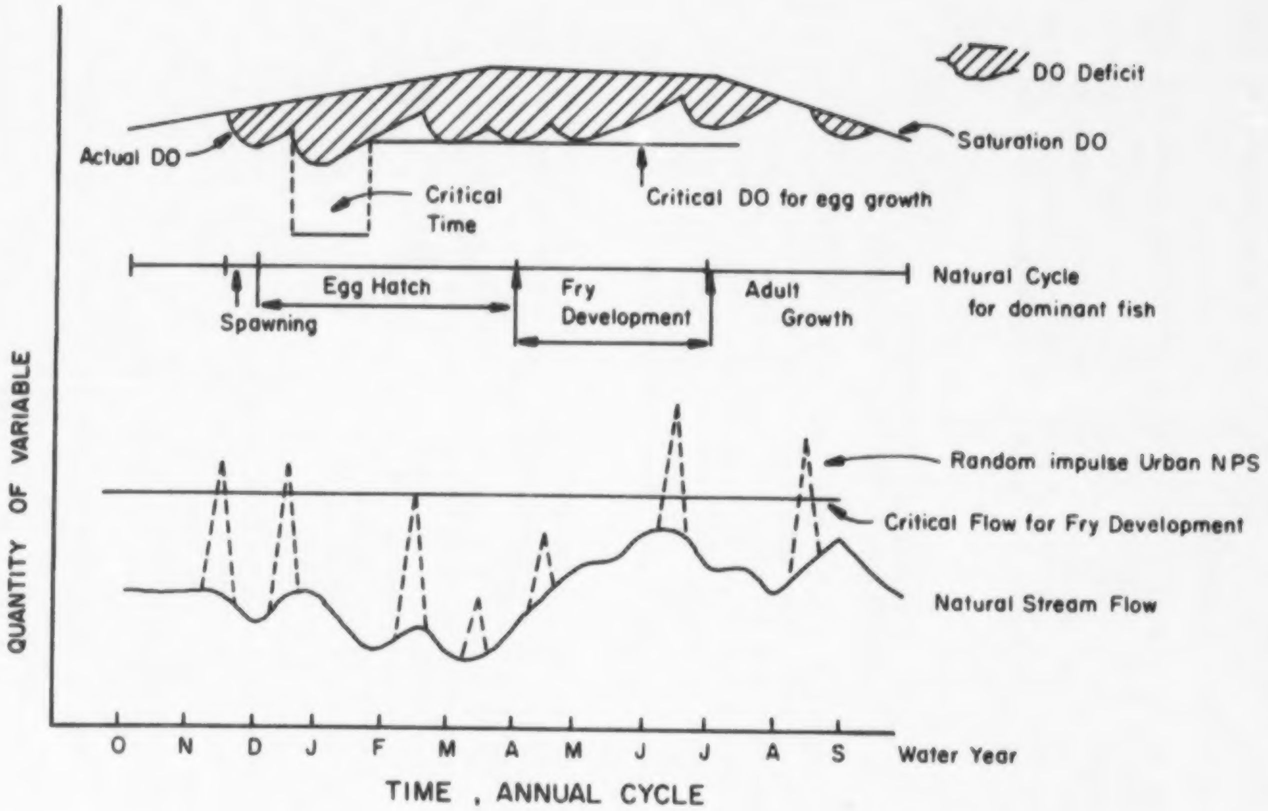


Figure 2. Hypothetical stream ecosystem effects of random impulse (urban NPS) loadings on ecological variables.

stream velocities to catch food and maintain safety from predation by large fish. If impulse loadings drive DO and flow variables outside acceptable ranges for a sufficient time and magnitude, a year class could be eliminated. Continued annual cycles where violations occur could eliminate the species.

The probability that a species or individual will not survive is equal to the probability that at least one event will occur that will eliminate the species or individual. For example, where flow is the event that threatens the species:

$$\text{prob (species eliminated)} = 1 - \text{prob } (Q \leq q_c) = 1 - F_Q\ (q_c) \qquad (1)$$

in which Q = flow

q_c = critical value of flow above which species is eliminated

$F_Q(q_c)$ = cumulative probability distribution of flow evaluated at q_c

However, flow has a deterministic seasonal component and therefore, F_Q and Q can be written as functions of time. Also, the critical value of flow varies with the stage of development of the species and is also a function of time. Therefore, Equation 1 can be rewritten as follows:

$$\begin{aligned}\text{prob (species eliminated)} &= 1 - \text{prob } (Q(t) \leq (q_c(t)) \\ &= 1 - F_Q\ (q_c(t);\ t)\end{aligned}$$

Similar relationships can be derived for other response variables. This approach is only a flow magnitude event, but this basic procedure can be extended to the bivariate case of a flow magnitude-duration event system. With the relationship developed above, survival versus probability curves could be obtained that relate to inputs having a certain probability (Figure 3). Similarly, the approach could be extended to variables that represent ecosystem processes, cycling and energy flow.

The effects of a certain urban NPS runoff input on a specific ecological variable would not be a single number but would vary within the probability of response depending on the space wherein the impact occurs (Figure 4)).

Because response is concentration related and high concentrations have a low probability of occurring, the probability of a species surviving or not surviving may be difficult to assess by modeling and calculation. Microecosystems could be used to develop a model and then be field tested. One means of testing this approach is to carefully measure the pattern of urban NPS inputs to a stream, determine the immediate response, relate the response to average (peak, range) concentration, and determine if the community structure

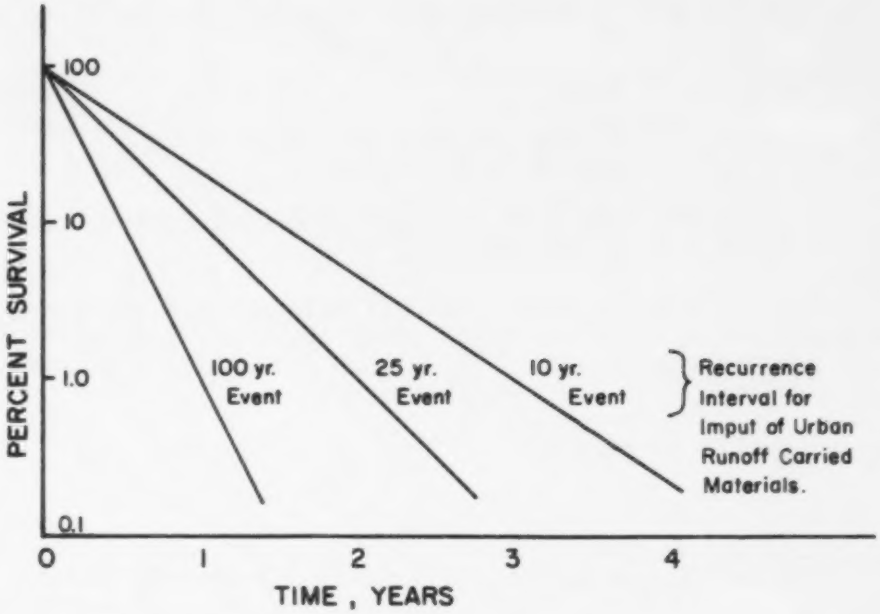


Figure 3. Percent survival decreases with more severe urban runoff events.

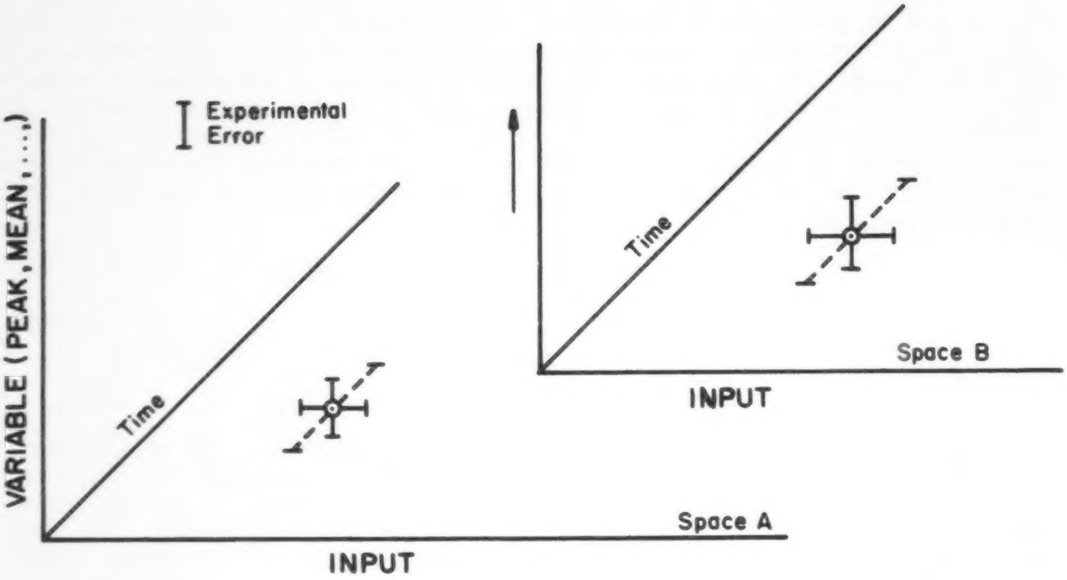


Figure 4. There is experimental error for system variables which adds to the variance caused by response to stochastic inputs.

and function varies with time (before/after) or space (upstream/downstream) as would be expected. The hypothesis for testing is: the community structure and function is more complex (or greater) than would be expected or than a control for a chronic input of material of the same total loading; i.e., random event and magnitude impulse loading is less deleterious than a chronic, steady loading.

Ecological variables for assessing these impacts would include time and space measurements of structurally and functionally defined biomass, photosynthesis, respiration, and diversity. These measurements, species lists, and other biological results could be assessed using various cluster analyses. Carbon and oxygen would be measured. Values would be estimated using mass units and energy units. Water quality and material input measurements would be made, especially throughout the flood hydrograph. The number of required measurements for monitoring and impact analysis could be determined by using rarefaction procedures (Simberloff, 1978). Eventually, based on a probability defined by policy, management or law, the required stream assessment sampling in time and space could be defined on the basis of the ecological resource being protected.

ACKNOWLEDGMENT

We thank those who contributed to the literature review, especially, D.L. Sorensen. D.L. Bowles, Utah Water Research Laboratory, J.D. Dean and S.W. Zison of Tetra Tech, and G.S. Schuytema and K. Randolph of the Corvallis Environmental Research Laboratory (U.S. EPA) reviewed the paper. Funding was provided by the U.S. Environmental Protection Agency, Corvallis (Order No. CC81224-J).

REFERENCES CITED

Ball, R.C., N.R. Kevern, and T.A. Haines. 1973. *An ecological evaluation of stream eutrophication.* Tech. Rpt. #36. Inst. Wat. Res., Mich. St. Univ. East Lansing, MI.

Cairns, J., Jr. and K.L. Dickson. 1977. Recovery of streams from spills of hazardous materials. *In: Recovery and Restoration of Damaged Ecosystems.* J. Cairns, Jr., K.L. Dickson, and E.E. Herricks, eds. Charlottesville, VA. pp. 24-42.

Cairns, J., Jr., K.L. Dickson, and J.S. Crossman. 1970. The biological recovery of the Clinch River following a fly ash pond spill. *25th Industrial Waste Conference.* Part One. pp. 182-192.

Chisholm, J.L. and S.C. Downs. 1978. Stress and recovery of aquatic organisms as related to highway construction along Turtle Creek, Boone County, West Virginia. *Geol. Sur. Wat.-Supply Paper 2055.* U.S. GPO, Wash., D.C. 20402. 40 p.

Cornell, H., L.E. Hurd, and V.A. Lotrich. 1976. A measure of response to perturbation used to assess structural change in some polluted and unpolluted stream fish communities. *Oecologia (Berl.)* 23:335-342.

DiGiano, F.A., R.A. Coler, R.C. Dahiga, and B.B. Berger. 1975. A projection of pollutional effects of urban runoff in the Green River, Massachusetts. *In: Urbanization and Water Quality Control.* Annual Symp. of the Amer. Wat. Res. Assoc. Proc. 20. pp. 28-37.

DiGiano, F.A., R.A. Coler, R. Dahiga, and B.B. Berger. 1976. *Characterization of urban runoff - Greenfield, Massachusetts. Phase II.* Wat. Res. Res. Cent., Univ. of Mass. Amherst, MA. 137 p.

Harrell, H.L. 1978. Response of the Devil's River (Texas) fish community to flooding. *Copia.* 1:60-68.

Krishnan, K., P. Radha, J.J. Lizcano, L.E. Erickson, and L.T. Fan. 1974. Evaluation of methods for estimating stream water quality parameters in a transient model for stochastic data. *Wat. Res. Bull.* 10:899-913.

Morse, W.L. 1978. The dishonest method in stream temperature modeling. *Wat. Res. Res.* 14:45-51.

Orians, G.H. 1975. Diversity, stability and maturity in natural ecosystems. *Unifying Concepts in Ecology.* W.H. Van Dobben and R.H. Lowe-McConnell, Eds. pp. 139-150.

Padgett, W.J., G. Schultz, and C.P. Tsokos. 1977. A stochastic model for BOD and DO in streams when pollutants are discharged over a continuous stretch. *Intern. J. Environ. Studies.* 11:45-55.

Padmanabhan, G. and J.W. Delleur. 1978. *Statistical and stochastic analyses of synthetically generated urban drainage quantity and quality data.* Tech. Rept. No. 108. Wat. Res. Cen., Purdue Univ. West Lafayette, IN. 111 p.

Porcella, D.B. and D.L. Sorensen. 1979. Characteristics of nonpoint source urban runoff and its effects on stream ecosystems. Report to U.S. EPA, Corvallis, OR 97330. 99 p.

Ragan, R.M. and A.J. Dieteman. 1975. Impact of urban stormwater runoff on stream quality. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc., Proc. 20.* pp. 55-61.

Rickert, D.A., W.G. Hines, and S.W. McKenzie. 1975. Planning implication of dissolved oxygen depletion in the Willamette River, Oregon. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc. Proc. 20.* pp. 70-84.

Simberloff, D. 1978. Use of rarefaction and related methods in ecology. *In: Biological Data in Water Pollution Assessment: Quantitative and Statistical Analyses.* K.L. Dickson, J. Cairns, Jr., and R.J. Livingston, Eds. ASTM. Philadelphia, PA 19103. pp. 150-165.

Slatkin, M. 1978. The dynamics of a population in a Markovian environment. *Ecol.* 59:249-256.

Slobodkin, L.B. and H.L. Sanders. 1969. On the contribution of environmental predictability to species diversity. *Brookhaven Symp. Biol.* 22:82-95.

Slocomb, J. and K.L. Dickson. 1978. Estimating the total number of species in a biological community. *In: Biological Data in Water Pollution Assessment: Quantitative and Statistical Analyses.* K.L. Dickson, J. Cairns, Jr., and R.J. Livingston, Eds. STP 652. ASTM, Philadelphia, PA 19103. pp. 38-52.

5

Fourth Session

IMPACTS ON RIVERS

Moderator: Robert Shubinsky
Water Resources Engineers
Springfield, Virginia

ANALYSIS OF RECEIVING STREAM IMPACTS ON THE MILWAUKEE RIVER

Thomas L. Meinholz
EcolSciences, Inc.
Milwaukee, Wisconsin

ABSTRACT

The Milwaudee Metropolitan area contains approximately 15,000 acres of combined sewers which discharge to the three rivers of the area or directly to Lake Michigan. As part of the Facilities Planning efforts of the Milwaukee Metropolitan Sewerage District and research for EPA's Storm and Combined Sewer Section, extensive monitoring and modeling efforts were performed to quantify the receiving water impacts of the combined sewers.

Over 100 overflow points within the combined system were modeled using the EPA SWMM model and Corpos of Engineers STORM model. The output of these models was used to load Harper's receiving water model. Major difficulties in modeling the river systems were the variable influence of Lake Michigan in the lower reaches. The relatively clean, high DO and lower temperature lake inflows could not be easily quantified because of the dependence of the inflows on wind speed, direction, lake level and other variable factors. Final model calibration using a linearly decreasing flux was found to match the continuous DO data generated during two years of record.

The Milwaukee river exhibits tremendous DO sags in the lower reaches of the river following runoff events. The modeling tasks of the CSO project could not duplicate these sags using the instream concentrations found in the monitoring program. Extensive field monitoring was then conducted to quantify the source and mechanism of these sags. After numerous investigations, the bottom sediments in the lower reaches were found to be the source of the rapid DO declines. The mechanism was related to the scouring action of submerged combined sewer outfalls.

In order to model the response of the river to discharge events, the receiving water model was modified to include an expression which would predict the extent and duration of the scour action from the submerged outfalls. Long term simulations of DO and other parameters were calibrated and verified using the response of the river to a multitude of rainfall events.

The use of this model network in the evaluation of alternatives for abating combined sewer overflows produced magnitudes of DO and fecal coliform impacts for each alternative using 20 years of rainfall record. The results for the following alternatives are presented.

- existing conditions
- partial separation
- complete separation
- out-of-basin (storage-conveyance-treatment)
- end-of-pipe (EOP)
- 100% CSO removal

Discussions of how these results were quantified including the model development will be discussed. The cost-benefit analyses that were used to satisfy the EPA PG-61 requirements will also be detailed to provide the reader with a methodology that has been successfully applied and approved in the Milwaukee area.

STUDY AREA

The Milwaukee area contains approximately 15,000 acres (6060 hectares) of combined sewers that discharge to the three rivers of the area which flow into Lake Michigan as shown in Figure 1. In 1975 the Milwaukee Metropolitan Sewerage District initiated a project to identify alternatives to abate combined sewer overflow (CSO) and select the alternative which was most cost-effective in terms of the stated water quality goals for the area. The monitoring and modeling investigations of this CSO project identified significant receiving water impacts as a result of wet weather discharges.

The preliminary findings of the CSO project led EPA to select Milwaukee as one site for another project to quantify the magnitude of the receiving water impacts associated with CSO. The Milwaukee area was selected as one of the sites for the impact evaluation project because of its large combined sewer area, the availability of previous and ongoing projects relating to CSO, and the significant water quality impacts which occur as a result of wet weather discharges. Each of the rivers has an associated combined and storm sewer area which discharges during wet weather. For purposes of this study, the investigations into the water quality impacts were limited to the Milwaukee River.

The most dominant feature of the river is the inflow from Lake Michigan in to the lower reaches of the river which influences flow conditions as far upstream as the North Avenue Dam (Figure 2). The inflow of lake water results in the slowing or reversal of flow in these reaches throughout the year. The lake influence has been monitored at a few downstream locations with a sensitive current speed and direction meter. The findings of these surveys have shown the river to be flowing upstream at one depth in the water column and downstream at another depth with a complete reversal of this trend a few moments later. Attempts to relate the inflows to wind speed, river flow and lake level have proved fruitless.

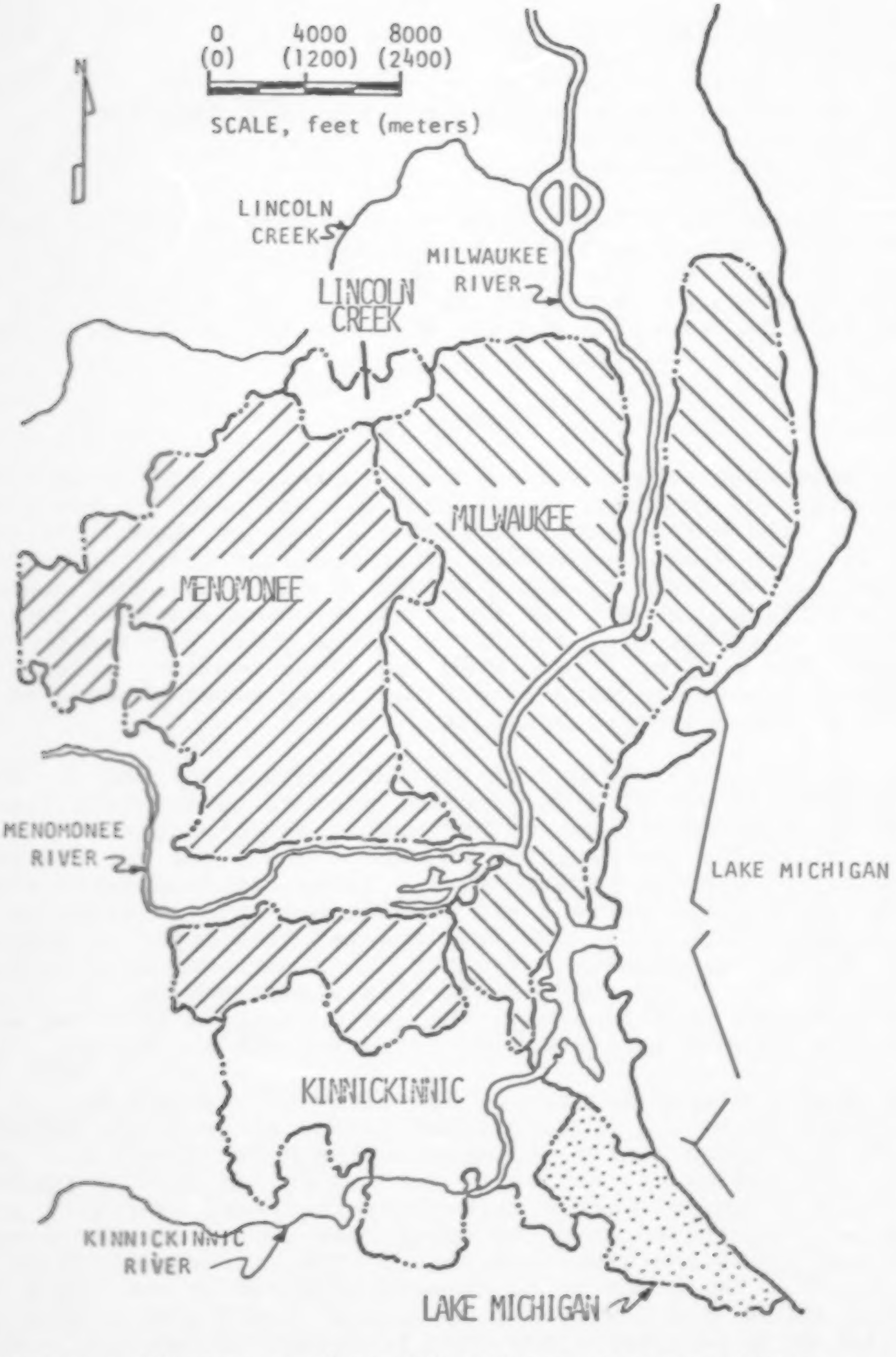


Figure 1. Illustration of the CSO drainage areas contributing to the various receiving waters in Milwaukee, Wisconsin.

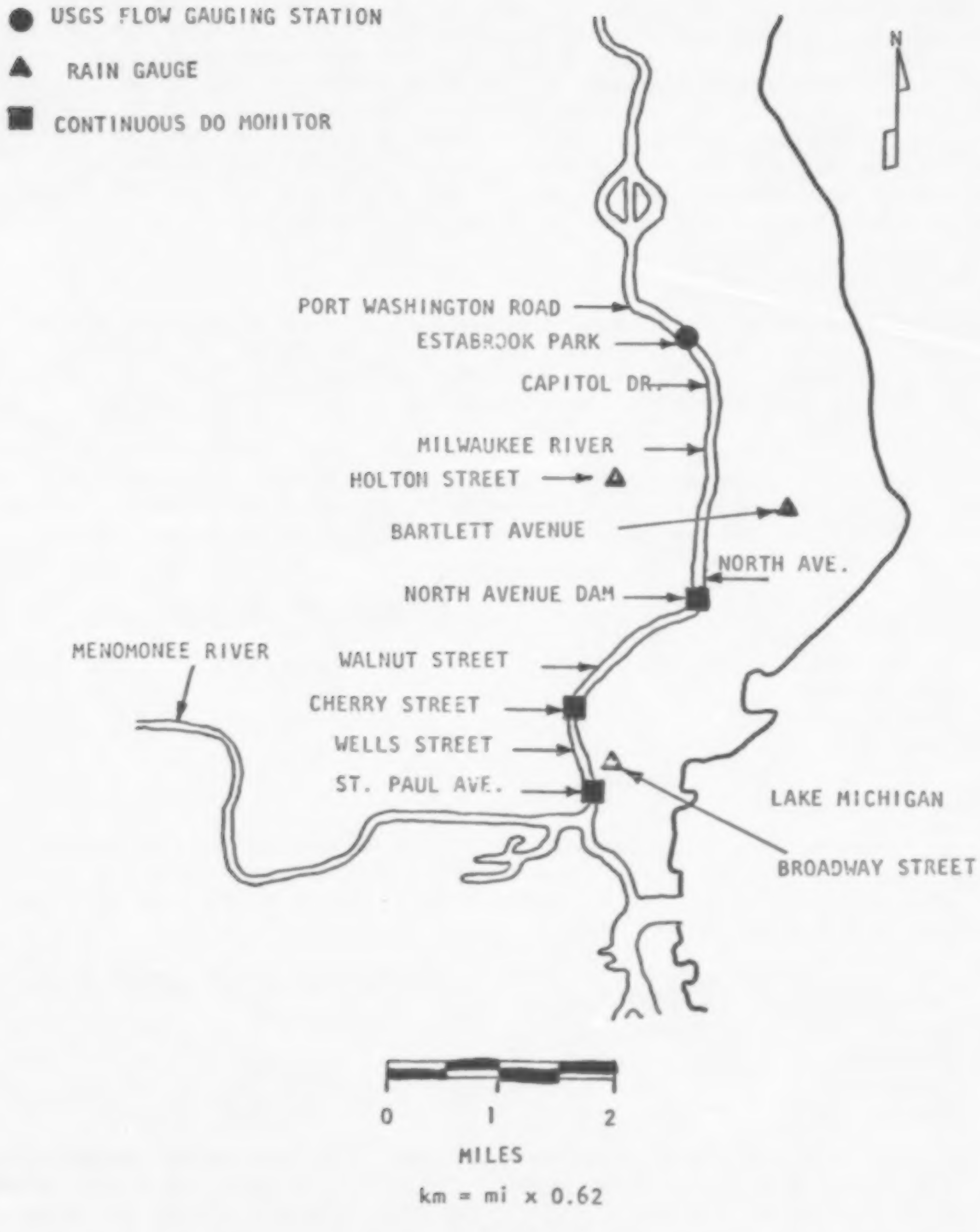


Figure 2. Monitoring locations in the study area of the Milwaukee River.

The mean annual discharge for the river as measured by the USGS gauge at Estabrook Park is 400 cfs (11.3 m^3/sec). During the coldest portion of the winter months, the river is ice covered in both the lower and upper reaches. The CSO area that discharges to the river extends along both sides of the river with the first outfall located at Capitol Drive as shown in Figure 2. A majority of the outfalls in the lower portions of the river are completely or partially submerged. Typical cross-sections of the river at a downstream location and a photograph of the lower reach site at St. Paul Ave. is shown in Figure 3.

Combined Sewer Area

The combined sewer area that contributes to the Milwaukee River is approximately 6000 ac (2428 ha) with a total of 52 outfalls ranging in size from 12 in. (30.5 cm) to a double 10 by 7.5 ft (3 by 2.3 m) box outfall. The drainage areas for individual outfalls range in size from 5 to 702 ac (2 to 284 ha). The entire combined sewer area has been modeled in a previous project using the EPA Storm Water Management Model (SWMM) and the Army Corps of Engineers Storage Treatment Overflow Runoff Model (STORM) (II). The land use percentages of the CSO area from the model data are as follows:

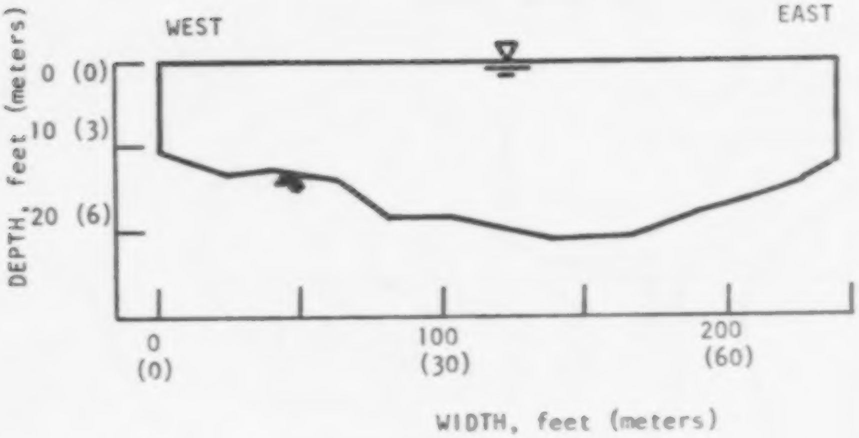
Land use	Percent of CSO area
Single family residential	57.6
Multi family residential	12.6
Commercial	17.8
Industrial	11.3
Parkland	0.7

Storm Sewer Area

Those areas within Milwaukee County that drain to the Milwaukee River and are served by storm sewers comprise approximately 27,000 ac (10,900 ha) of drainage area. These areas contain the following land use breakdown:

Land use	Percent of storm sewer area
Single family residential	50.1
Multi family residential	2.7
Commercial	12.9
Industrial	4.4
Park - open area	29.9

This area has also been modeled with the SWMM and STORM models. Figure 4 illustrates a comparison of the storm sewer and CSO areas. Some of the storm sewers within this area contain cross connections with the sanitary sewers so that during wet weather the surcharged sanitary system may be relieved through nearby storm sewers.



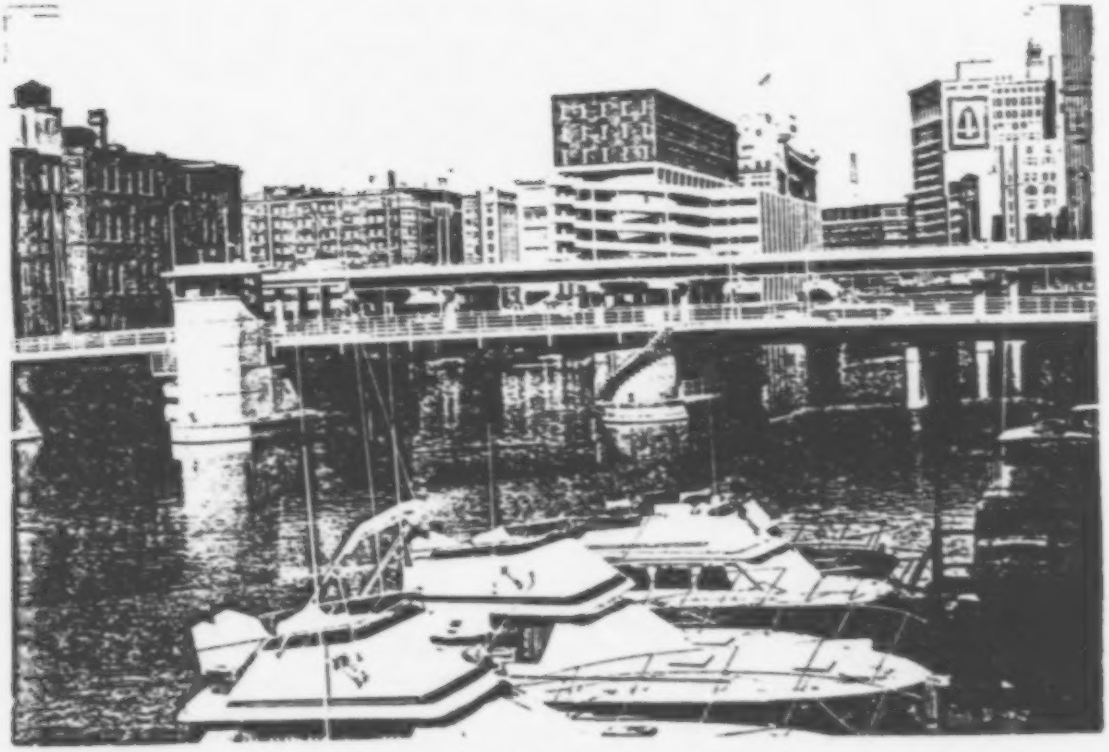


Figure 3. Photograph and cross section of the Milwaukee River at St. Paul Avenue

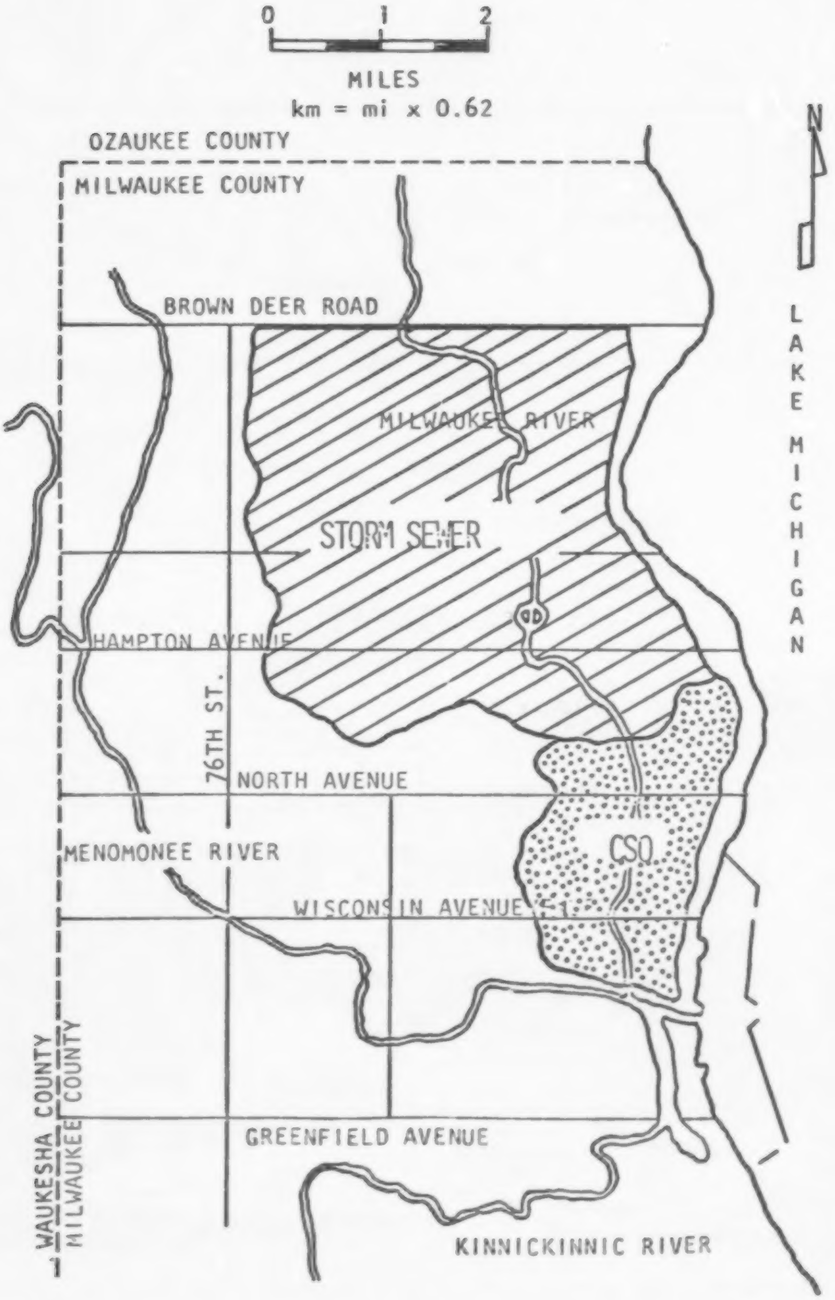


Figure 4 Illustration of CSO and storm sewer areas tributary to the Milwaukee River.

Upstream Areas

The Milwaukee River watershed is comprised of 644 mi^2 (1668 km^2) of drainage area that lies to the north of the Milwaukee metro area. The predominant land use within this area is agricultural in nature which comprises approximately 66 percent of the upstream watershed. Urban land use accounts for 1.5 percent while open areas and woodlands make up the remaining area. In 1970, an estimated 530,000 persons resided in the entire watershed including the area within Milwaukee County.

FIELD INVESTIGATIONS

The water quality conditions within the Milwaukee River exhibit extreme variations between wet and dry weather periods. Figure 5 presents data from a continuous DO monitoring device located at St. Paul Avenue showing the variation in DO. Figure 6 presents fecal coliform data from the same site during two intensive monitoring surveys. The dramatic loss of dissolved oxygen and rapid increases in fecal coliform concentrations were investigated in the field monitoring portions of the projects. These consisted of comprehensive field surveys, the analysis of two years of data taken from three continuous DO and temperature monitors, dye studies to define the rivers hydraulic and mixing characteristics, and sediment investigations.

The term water quality impact for this study was limited to the violations of the DO and fecal coliform standards that have been set by the Wisconsin Department of Natural Resources for the Milwaukee River. The standards apply for both dry and wet weather flow conditions in the river and are as follows:

Upstream of North Avenue Dam

1. Preservation and enhancement of fish and other aquatic life. The DO content shall not be lowered to less than 5 mg/l at any time.

2. Full body contact recreational use; the membrane filter fecal coliform count (MFFCC) shall not exceed 200 per 100 ml as a geometric mean based on not less than five samples per month.

Downstream of North Avenue Dam

1. Marginal conditions for fish and aquatic life. The DO shall not be lowered to less than 2 mg/l at any time.

2. Partial body contact recreational use; the MFFCC shall not exceed 1000 per 100 ml as a geometric mean based on not less than five samples per month.

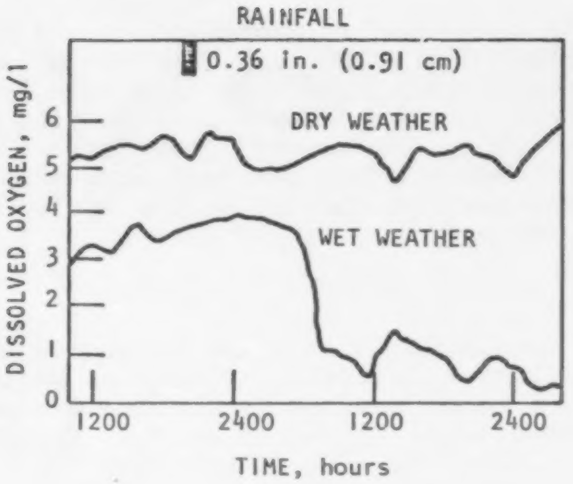


Figure 5. Illustration of dry weather (July 29-31, 1977) and wet weather (August 3-5, 1977) DO levels at St. Paul Avenue in the Milwaukee River.

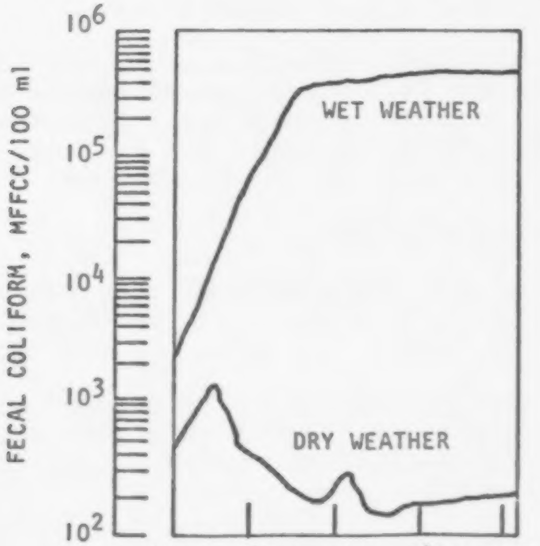


Figure 6. Illustration of dry weather (September 22-24, 1976) and wet weather (August 4-6, 1977) fecal coliforms levels at St. Paul Avenue in the Milwaukee River.

The difference in standards between the upper and lower portions of the river reflect the physical and biological characteristics of these two reaches. The upper portions above the dam are free flowing and not under the influence of Lake Michigan while the lower portions are those reaches which exhibit some degree of inflow from the lake.

Source of Impacts

The data collected in the monitoring portions of both projects were used in Harper's Multi-Parametric Model to attempt a DO calibration to determine the source of these impacts. An acceptable wet weather calibration for DO could not be made, however. Extensive work in modifying the model provided no better results. The magnitude and the duration of the predicted sags as well as the location was consistantly underestimated. At this time the bottom sediments in the lower reaches of the river were thought to be the possible source of these sags.

Sediment Oxygen Demand Tests

In order to determine the significance of the sediments on the dissolved oxygen balance in the Milwaukee River, both *in situ* measurements and laboratory testing was conducted. A respirometer consisting of a plexiglass chamber with a DO probe attached inside and a small 12 v submersible pump attached outside for circulation of water was used to determine the decline in DO over time after the chamber was sealed on the sediment surface. The *in situ* SOD rate was calculated using the following formula:

$$SOD = \frac{(Ci-Cf)V}{tA}$$

Where:

SOD = Sediment uptake rate in gm O_2/m^2-day
V = Volume of confined water in m^3.
A = Bottom area within chamber in m^2.
t = Test period in days.
C_i = Initial measured DO in chamber in mg/l.
C_f = Final measured DO in chamber in mg/l.

This formula provides the SOD rate on an areal basis and was converted to a volumetric rate in the river by division with the mean depth at the sampling location.

The results of the *in situ* SOD measurements showed the rates to vary from zero at the upper limits of the study area to an average of 5.0 gms/m^2/day in the lake influenced reaches. The new values were utilized in the DO sink equations of Harper's model with the output as follows;

1) accurate DO predictions during dry weather at all stations
2) accurate DO predictions during wet weather at upstream (non-lake influenced locations

3) poor prediction of DO magnitudes during wet weather at downstream locations.

These findings led the project team to further investigate the oxygen demand of the bottom materials in the laboratory to determine the mechanism of the dramatic DO sags.

Samples of the bottom materials were collected and placed in plexiglass chambers in the laboratory. River water was placed over the sediments and allowed to settle. After lightly aerating the river water, the rate of decline of the DO was compared with the corresponding *in situ* measurements.

In most cases the laboratory measurements were slightly lower than the field SOD's which indicated the bottom materials may have been slightly disturbed when the SOD chamber was lowered into the sediment. This phenomenon led the project team to investigate the disturbed demand of these sediments to quantify the change in SOD magnitude. The results of these investigations are shown in Table 1 and Figure 7. Note that the disturbed demand shows more than a thousand fold increase in SOD over undisturbed conditions. This disturbed or scoured demand was thus, identified as the source of the extensive DO sags.

Mechanism of Impact

The determination of the disturbed SOD as the source of the DO impact next led to the identification of the mechanism of the impact. Questions such as what causes the sediments to be disturbed (scoured) needed to be answered so that the receiving stream model could be properly calibrated. Settling tests of these sediments indicated the scouring velocity must exceed 0.1 foot per second (.03 m/sec). A program of velocity monitoring at the sediment river interface was then undertaken in the river during different flow conditions. The locations selected for the monitoring were those that had shown significant DO sags during previous storms. A sensitive velocity meter was lowered to approximately 1 foot (.3m) of the bottom of the twenty foot water column. Velocities were continuously monitored to determine if scouring velocities were present.

The scour of bottom materials within the river by discharges from submerged CSO outfalls was verified with the velocity studies and with visual observations. The velocities measured in the vicinity of the submerged outfalls were extremely large and capable of scouring considerable quantities of sediment. The effect of the

TABLE 1. BENCH SCALE DETERMINATIONS OF SEDIMENT OXYGEN DEMAND UNDER UNDISTURBED AND DISTURBED CONDITIONS[1].

Map location no.	Description	Undisturbed SOD, gm/m^2-day	Disturbed SOD, gm/m^2-day
1	Kinnickinnic River at First Street	0.65	430
2	Mooring Basin in Inner Harbor	1.40	1,370
3	Milwaukee River at RR Bridge	2.10	800
4	Menomonee River at Great Lakes Coal	1.70	270
5	Milwaukee River at Highland Avenue	1.40	360
6	Milwaukee River at Hubbard Park	0.33	66

[1] The laboratory sediment oxygen demand determinations were carried out at a temperature of 20 ± 0.5 C.

scouring can be illustrated with the rainfall, DO and CSO velocity data for a storm occurring on August 3, 1977. Figure 8 depicts the response of the river at St. Paul Avenue in terms of DO to this rainfall event and the subsequent scouring of sediments due to CSO discharges. An instream CSO velocity of nearly 12.0 ft/sec (3.7 m/sec) was measured 1.0 ft (0.3 m) above the sediment surface at a distance of approximately 30 ft (9.1 m) from a submerged CSO outfall at St. Paul Avenue. This velocity was measured just after discharges from this CSO outfall began and the direction of the velocity was perpendicular to the river. After discharge from the outfall ceased, the river velocity was variable in direction and very small in magnitude. The rapid decline of DO shown in Figure 8 following the rainfall event is the result of the scouring or agitation of bottom sediments throughout the lower Milwaukee River from discharges from submerged CSO outfalls. This CSO velocity measurement, as well as other measurements indicate that sediment scouring from submerged CSO discharges does occur in the lower portion of the river.

Visual observations in the lower river have also indicated the significance of sediment scouring. During severe storm events, the plumes from the numerous outfalls and the resulting scour of sediments were observed. The overall significance of scouring from CSO discharges is evident when one considers that there are

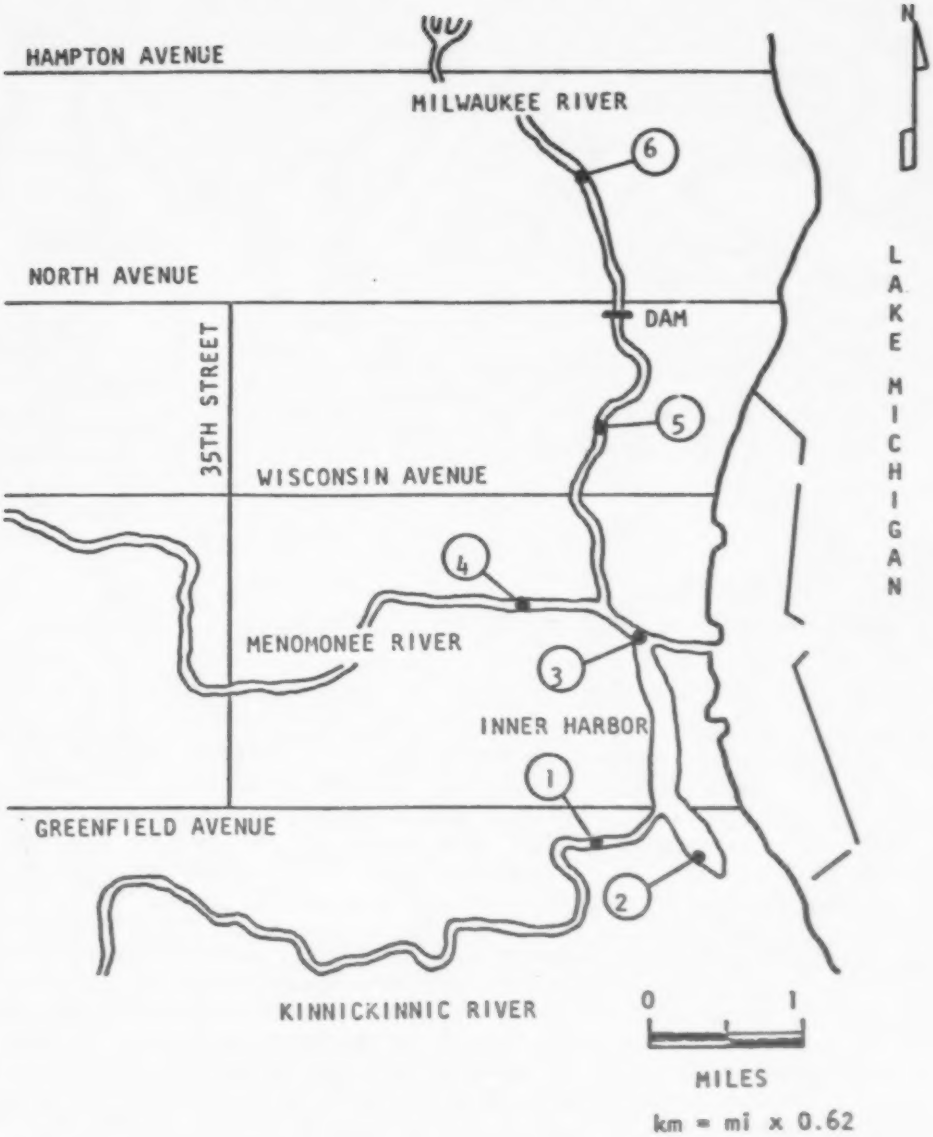


Figure 7. Sediment sampling locations for bench scale sediment oxygen demand characteristics.

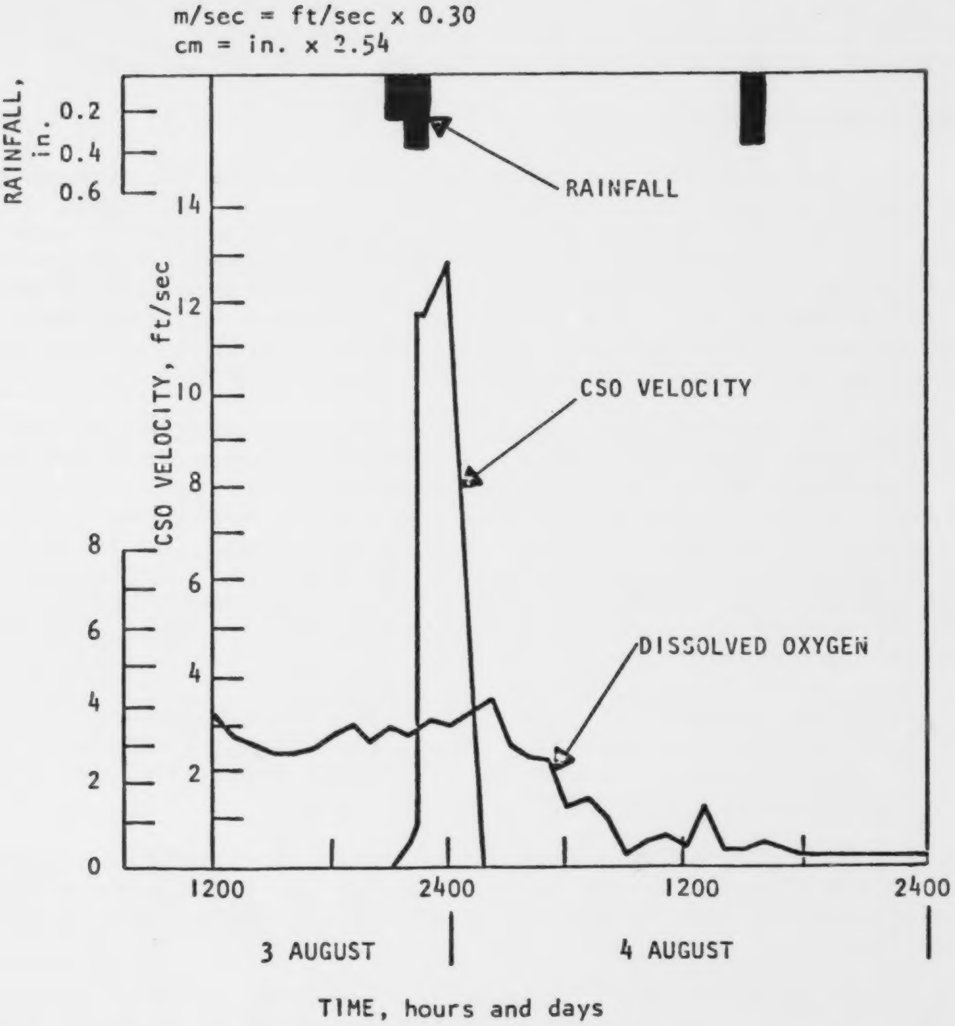


Figure 8. The observed decline in DO, instream velocity from a submerged CSO outfall, and rainfall volume at St. Paul Avenue in the lower Milwaukee River on August 3-4, 1977.

approximately 40 outfalls discharging into the Milwaukee River below the North Avenue Dam. Although some of these outfalls have irregular shapes, the majority are circular and range from 36 in. (0.9 m) to 96 in. (2.7 m) in diameter. Also, most of them are submerged. Averaging the number of outfalls evenly over the length of this portion of the river, approximately one CSO outfall exists every 300 ft (95 m) of river. Considering that many of the outfalls enter the river in pairs from opposite banks, and considering frequency of the outfall occurrence, the scouring potential from these outfalls can be quite significant.

Long Term Modeling

In order to evaluate the water quality effects of various CSO abatement alternatives, a long term simulation of water quality was devised to quantify the water quality improvement associated with each alternative. Harper's model was used to evaluate years of rainfall-runoff loads from the contributing areas as predicted by the STORM model. The output of the model was then used to determine the frequency or magnitude of standards violations in the river as a function of the CSO alternatives.

In order to predict the impact of sediment scouring by CSO outfalls during rainfall events an emperical expression was developed for the model using multiple regression analyses. Observed decreases in DO at the continuous monitoring locations in the river following 20 rainfall events of varying volume and intensity were analyzed versus the rainfall, runoff and river conditions for each event. The change in dissolved oxygen concentration (decline) was correlated with the following variables to determint the most significant relationship:

Rainfall volume
Peak rainfall invensity
Average rainfall intensity
Volume of CSO discharge
Peak river flow
Change in river flow
Time since previous runoff event

The results of this analysis indicated that rainfall volume, volume of CSO discharge and time since the previous events were significant at the 95 percent confidence limits using a linear equation.

A similar relationship has been developed to predict the duration of the decline in DO since this parameter is important in quantifying the frequency of DO violations. The duration is much more difficult to predict since the lower reaches of the river are influenced by the flux of water from Lake Michigan. Despite this difficulty, a linear relationship of DO sag duration to rainfall volume, CSO volume and time since the previous storm was found to be significant at the 90% confidence limits.

Within the long term simulation of water quality an expression has been added to the model to provide for a time varying sediment oxygen demand (SOD) rate. This SOD rate is modeled as an immediate high demand (disturbed) at the beginning of a CSO event which is reduced at an exponential rate to the dry weather (undisturbed) demand between storms. The magnitude and duration of the increase in SOD rates are determined through the use of previously described empirical equations. The equations for time varying SOD rates in the model are controlled by three coefficients:

Magnitude (A)
Decline (B)
Time of Decline (T)

The A Coefficient is used to adjust the magnitude of the initial increase in the SOD rate due to a particular storm event while the B coefficient is used to adjust the exponentional decline in this high SOD rate to the lower dry weather rate. Coefficient T is the total duration of the increased SOD rate as predicted with the regression equation.

Harper's model is quite sensitive to the A and B coefficients in terms of the rate of dissolved oxygen decline following a storm event. These coefficients can be adjusted in a manner that nearly all of the decline in dissolved oxygen occurs during the first or second 3-hour time step following a storm event. In order to match observed changes in the DO concentration in the lower Milwaukee River, the A and B coefficients are set so that the ratio of the peak SOD rate to average SOD rate following a storm ranges between 1.1 and 1.5 depending on the size of the storm event. Although this method of setting the coefficients is arbitrary, the predicted dissolved oxygen decline matches the observed decline at St. Paul with good accuracy.

The major assumption in the development of this procedure for predicting the DO impact of scoured sediments is that the observed rapid decline in DO is primarily due to the sediments. Some of the decline is due to the organic loading from CSO, storm sewers and upstream loads, however, the rates of DO decline far exceed the rates which could be due to the measured BOD_5 and TOC concentrations in the river. For this reason, the previous assumptions are justified and when implemented in the model, provide an excellent prediction of the monitored dissolved oxygen concentrations.

EVALUATION OF CSO IMPACT

In order to evaluate the contribution of CSO to the DO impacts within the Milwaukee River, a sensitivity analysis using the instream model network with various CSO loads was conducted. The STORM and instream model were run for one year of rainfall for the period April through October when non-frozen ground and non-ice cover conditions are present. In order to measure the differences in the DO curves that are computed by Harper's model, the technique shown in Figure 9 was utilized. The shaded area below 5.0 mg/l standard represents the magnitude of the dissolved oxygen impact which has the units of mg-day/l. The magnitude measurement represents the difference in water quality between a given standard and the predicted values. Another measurement which is used to quantify the DO impact is the number of days of violation that occur within the simulated time period. Thus, any part of a day which contains a violation of the standard will be listed as such. Figure 9 presents the data where there are eight DO values listed per day in the model output. The 5.0 mg/l standard is violated during day 2 and 3 of the period with day 2 having four values below the 5.0 mg/l standard and day 3 having one. Two days of violation will then be listed along with the magnitude of the violations for this period.

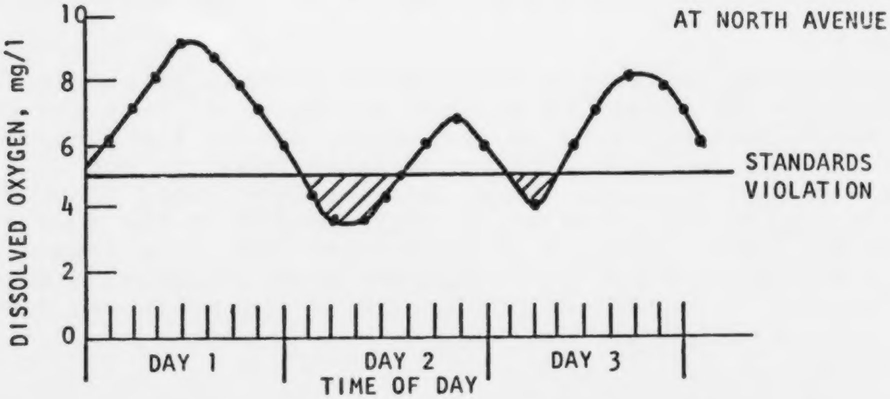


Figure 9. Typical model output for determining frequency and magnitude of dissolved oxygen violations.

In order to quantify the sensitivity of the river to the loads generated by CSO, an extreme year of rainfall was selected from the 38 years of record available for the Milwaukee area. The predicted flow and quality from STORM for each storm event were input to Harper's model to generate the frequency and magnitude of DO violations at each site for the April through October period using mean monthly low flows. The DO standards for determining the magnitude the frequency of violations were 5.0 mg/l for sites upstream of the North Avenue Dam and 2.0 mg/l for sites downstream of the dam as discussed previously.

Next, the predicted pounds of BOD, suspended solids and numbers of fecal coliforms for each overflow event during the year in the CSO area were doubled and loaded into Harper's model with the same storm sewer and upstream boundary loads as the previous model runs. This technique was used to determine the change in DO quality with this 200 percent increase in CSO pollutant load. Similarly, the CSO load and flow was also completely removed to simulate the complete elimination of CSO. These three simulations were run using the CSO sediment scour potential for each runoff event and they were run without the scour mechanism in order to quantify the differences between these conditions. The results of these model runs shown in Figure 10 for both the with and without scour mechanism.

The data of Figure 11 provide a means of evaluating the contribution of CSO to the impacts on DO at various locations within the Milwaukee River. For example, the North Avenue data shows that the doubling of the CSO load results in an increase of approximately 20 percent in the magnitude of the DO violations of the 5.0 mg/l standard. Removal of all CSO at this site results in a 21 percent decrease in the magnitude. Sediment scouring from submerged CSO outfalls does not occur in the portions of the river upstream of the North Avenue Dam. Therefore, the DO impacts at North Avenue are strictly a function of the wet weather loads and the low flow conditions that were used for this analysis. At Walnut Street, where sediment scouring does occur, the differences are more pronounced. Doubling the CSO load results in a 78 percent increase in the magnitude of the standard violations of 2.0 mg/l. Complete removal of the CSO load results in zero magnitude below the standard. The contribution of sediment scour to these violations, as determined using the without scour results, is approximately 69 percent. The existing magnitude of DO violations is 17.2 mg-days/l with scour. This decreases to 5.3 mg-days/l (69 percent decrease) when the scour mechanism is removed. At higher CSO loads (200 percent), the sediment contribution to the magnitude of DO violations is reduced to approximately 37 percent at Walnut Street. This reduction in influence of sediments is due to the higher dissolved and suspended loads utilized in the 200 percent CSO load simulation.

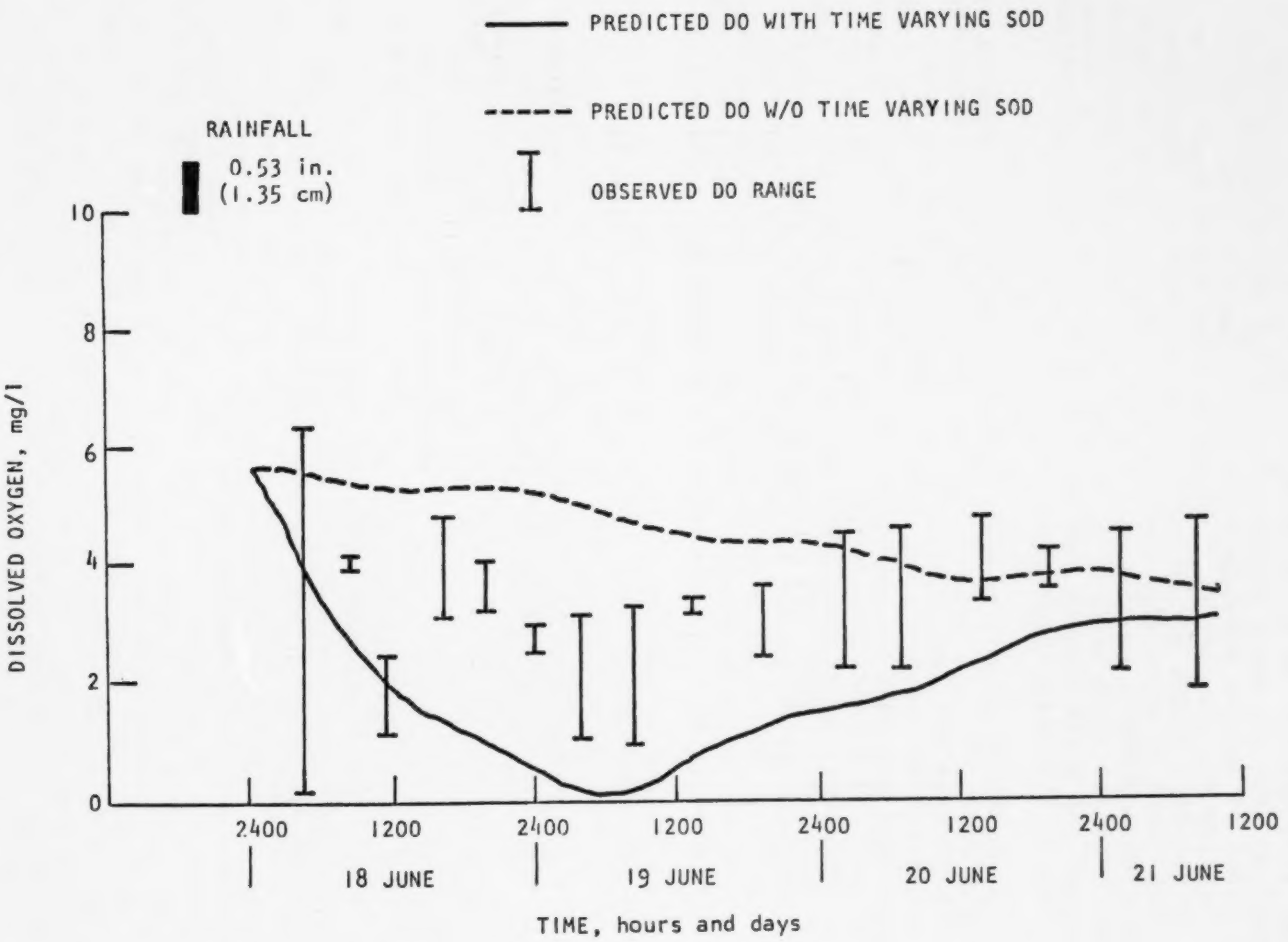


Figure 10. Wet weather calibration results for St. Paul Avenue, June 18-21, 1977 (Survey 3).

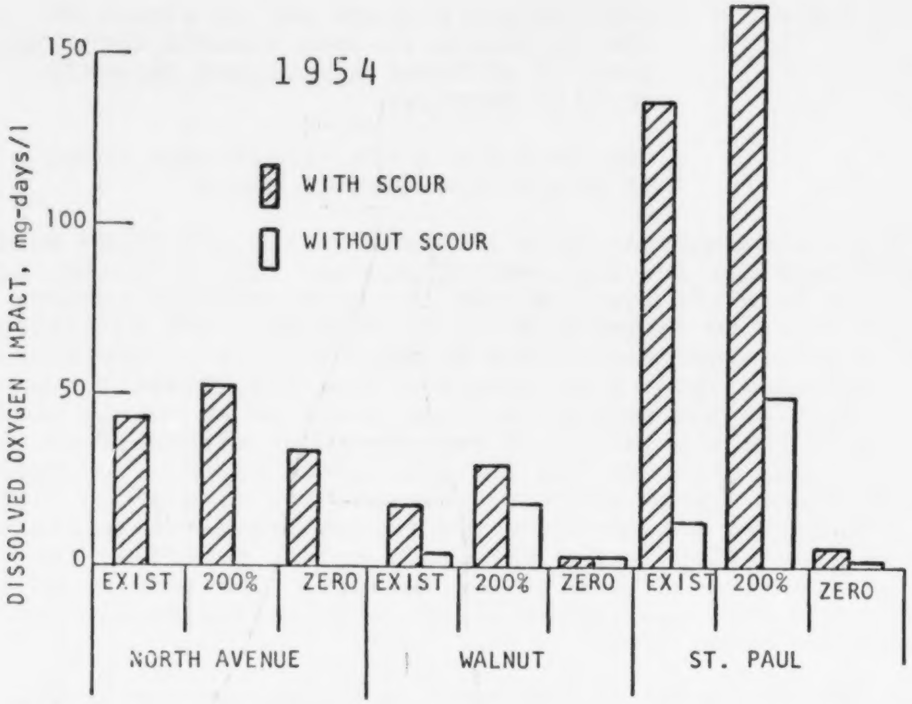


Figure 11. Sensitivity of dissolved oxygen results in CSO loads.

The St. Paul Avenue site has the most drastic changes in water quality when comparing the sensitivity of various loads. By doubling the CSO load, the DO magnitude below 2.0 mg/l increases by 22 percent while removing the CSO load reduces the impact by 96 percent. Removal of the scour mechanism from the model network reduces the impact of the existing CSO load by 91 percent when compared to the with scour alternative. This result indicates the significance of the scouring of sediments on DO conditions in the lower portions of the Milwaukee River. In comparison, the CSO loads have only a minor influence on DO levels during the low flow conditions used for this simulation.

Evaluation of CSO Abatement Alternatives

A total of three general CSO abatement alternatives were investigated using various pollutant removal efficiencies and design criteria. The three alternatives and their combinations are listed as:

Sewer separation	- partial and complete
End of Pipe	- 35% solids removal, 20% BOD removal, 98% fecal coliform removal

Out of Basin - conveyance, storage and treatment of the overflows in deep tunnels with the treated effluent discharged directly to Lake Michigan

- designed for a 1/2, 1,2,5 year level of protection (design storm)

Each of these alternatives were simulated using the STORM model with hourly rainfall for one year of record. The pollutant loadings for each rainfall event within the April through October period were input to Harper's model to determine the resulting DO and fecal coliform concentrations within the river. The storm sewer and upstream loads that resulted from the rainfall events were also input to the receiving water model on an hourly basis. In addition to the simulation of CSO abatement alternatives, the model network was also run for two additional cases: the existing, unabated CSO system the total elimination of all CSO flow and quality. These two schemes were run to determine the sensitivity of the receiving water to the CSO loads and to determine the "do-nothing" alternative or present day results. The results of these simulations are shown in Figure 11 for three sites within the Milwaukee river study area.

Figure 12 represents the DO and fecal coliform results at the North Avenue dam site which is only slightly affected by the sediment mechanisms which are present in the lower reaches of the river. As can be seen in this figure, the DO results show little difference between the existing CSO conditions and any of the alternative CSO control techniques including 100% removal of the CSO. This pattern is a result of the DO impact at this site being related to the dissolved load that is present in the river from upstream sources which is not changed by the removal of CSO. The North Avenue site has approximately 50% of the CSO drainage area discharging upstream of this location but the impact on DO from these discharges is not expressed at this site. The fecal coliform results shown in this figure reflect the removal of CSO in a much more dramatic manner than the DO results. Thus, the zero CSO alternative shows one day of violation which is a result of the upstream loads of fecal coliform while this site contains 40 days of violation when the existing CSO load is present. The sewer separation alternative also shows the difference in complete removal of the sanitary component in the combined system and the partial removal of this element.

Figures 13 and 14 represent the DO and fecal coliform results for two downstream sites on the Milwaukee river. Here, the differences in DO magnitudes are much more evident than at the upstream site because of the presence of the sediments and the influence of the scour mechanism is more dominant. Each of the DO results for an alternative can be evaluated by relating the degree of

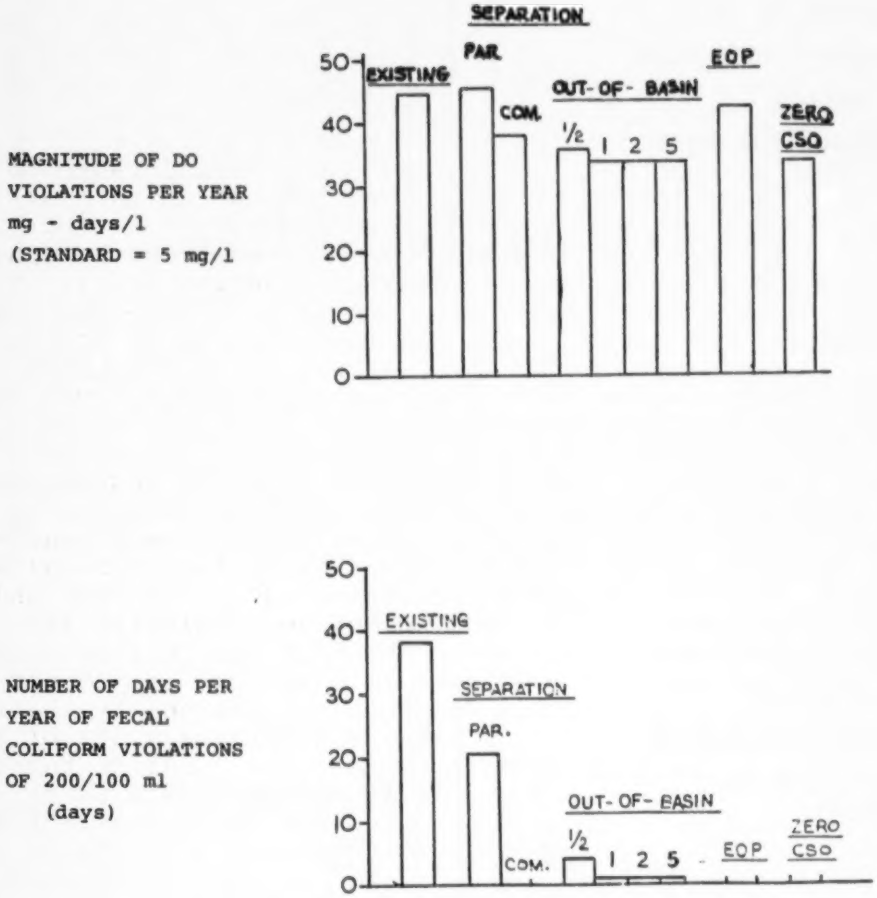
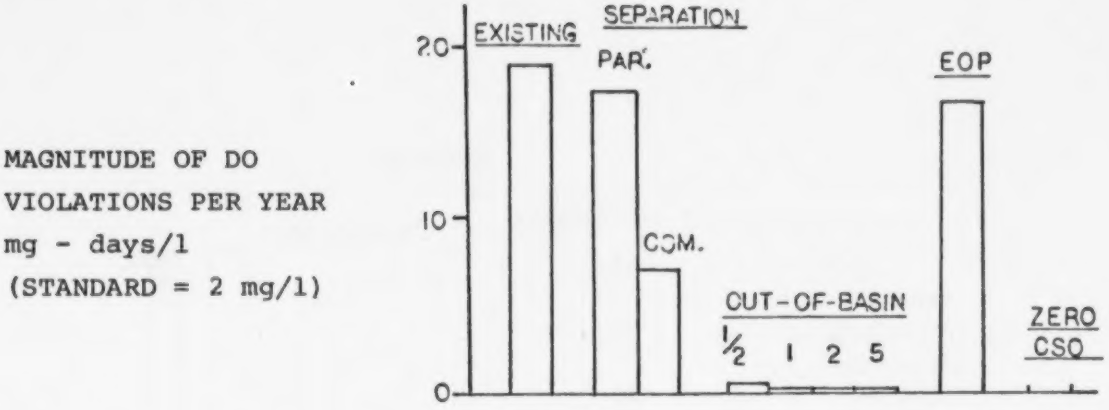


Figure 12. Comparison of the average dissolved oxygen and fecal coliform results during the typical and extreme runoff years for the alternatives at North Avenue.



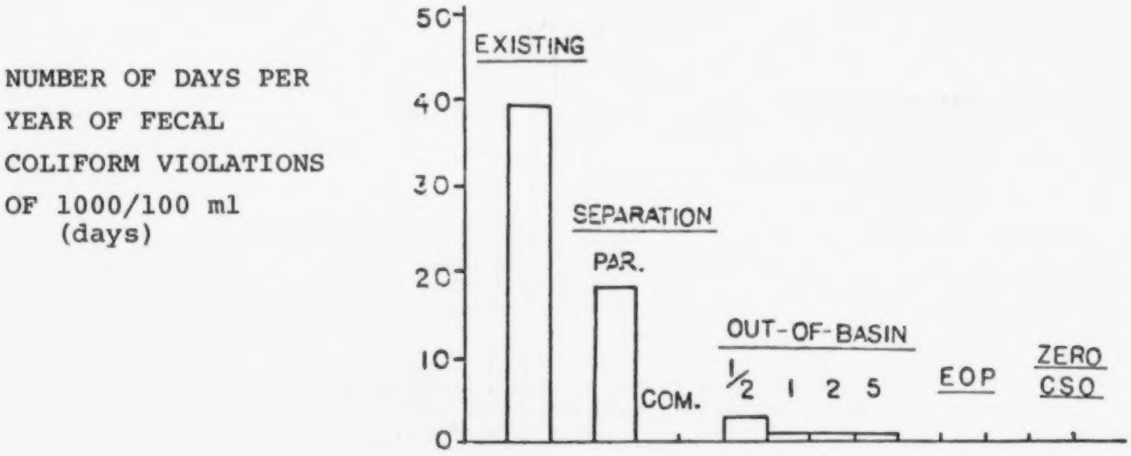


Figure 13. Comparison of the average dissolved oxygen and fecal coliform results during the typical and extreme runoff years for the alternatives at Walnut Street.

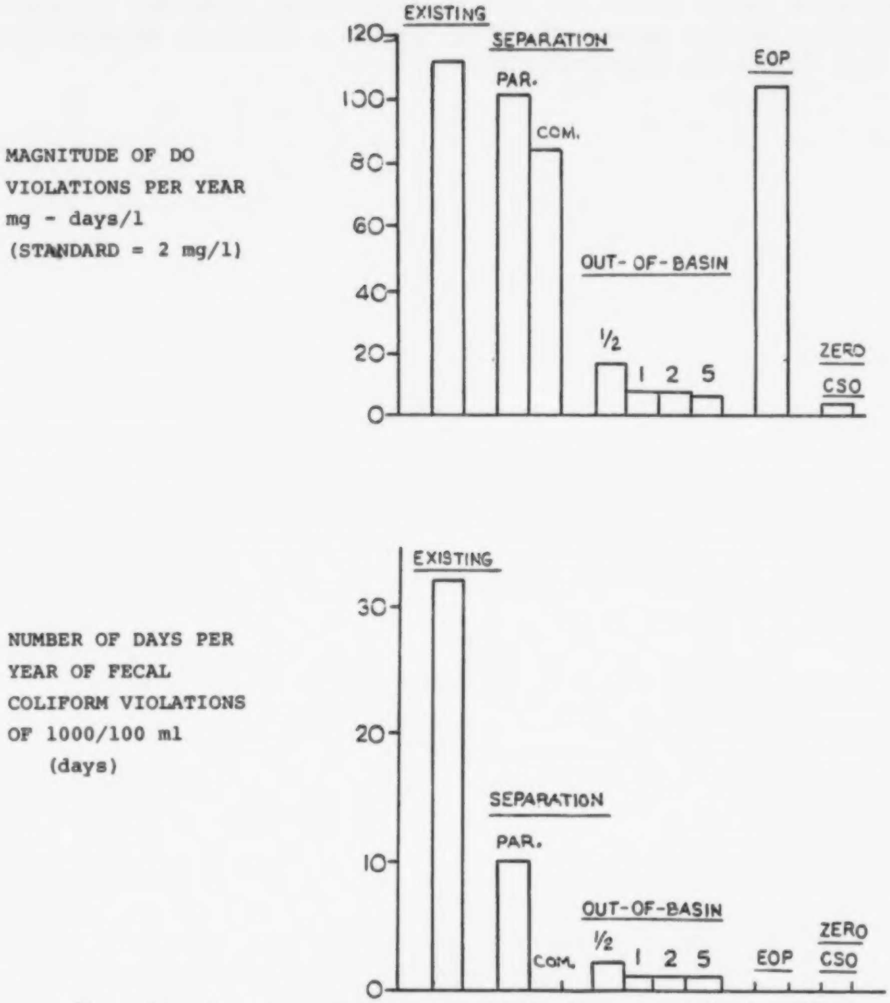


Figure 14. Comparison of the average dissolved oxygen and fecal coliform results during the typical and extreme runoff years for the alternatives at St. Paul Avenue.

scour associated with that alternative. For example, partial and complete separation do not remove the submerged outfalls, and as a result the scour potential of this alternative is the same as for existing conditions. The out-of-basin alternative removes the scour mechanism by routing the overflows to tunnels which eventually discharge outside the river basin. None of the alternatives provide the improvement in DO violations which could be achieved by complete elimination of CSO. However, the one, two and five year sizes of the out-of-basin concept provide DO improvements which approach this level. Partial separation provides the least DO improvement.

COST-EFFECTIVENESS OF ALTERNATIVES

The evaluation of each alternative in terms of water quality improvements cannot be completed until the cost associated with each is compared. The detailed costs required for this analysis are continuously being updated for the Milwaukee Metropolitan Sewerage District. It must be emphasized that the costs shown in the following pages are previous estimates and have been updated a number of times since this project was completed.

In order to provide an indication of the shape of the cost versus water quality improvement curve, cost data developed for the entire CSO area for the out-of-basin and sewer separation concepts was utilized. These cost data are presented on an annual basis in order to determine where the "knee" of the curve will occur. The total costs for all three river basins will be utilized throughout this analysis with the water quality data from the Milwaukee River to insure uniformity between in and out-of-basin alternatives.

Water Quality Improvement Versus Cost

The water quality improvement parameter for use in this analysis was quantified utilizing the percent improvement in DO magnitude or days of fecal coliform violations when compared to existing conditions. For the purposes of the following discussion, the extreme year of 1941 will be used for the examples in the text. The data to be used in developing the curve of cost versus water quality as measured for DO is shown in Figure 15. As the figure indicates, the four levels of protection are bunched at or near the zero CSO discharge line while the existing CSO conditions are at the other extreme with no points between. In order to place a line between the one-half year point and the existing conditions location a few assumptions must be made until further data is available. The first and most critical assumption is that a significant number of dollars on an alternative would have to be spent before any noticeable water quality improvements would occur. This means the line connecting the two points will start out from the existing conditions location (zero dollars/year) in an almost vertical trajectory before leveling off as it approaches the mid-point of the axis.

Figure 16 illustrates the DO and fecal coliform measurements of water quality improvement versus cost for the alternatives at St. Paul Avenue. The locations of the partial and complete sewer sep-

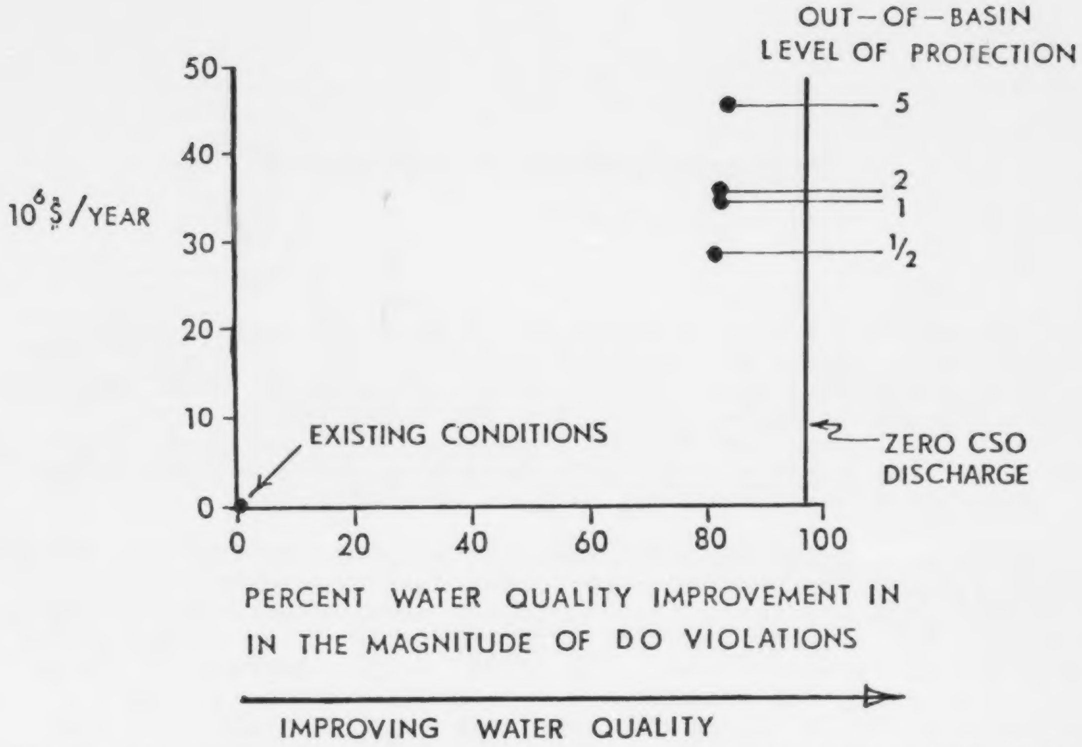
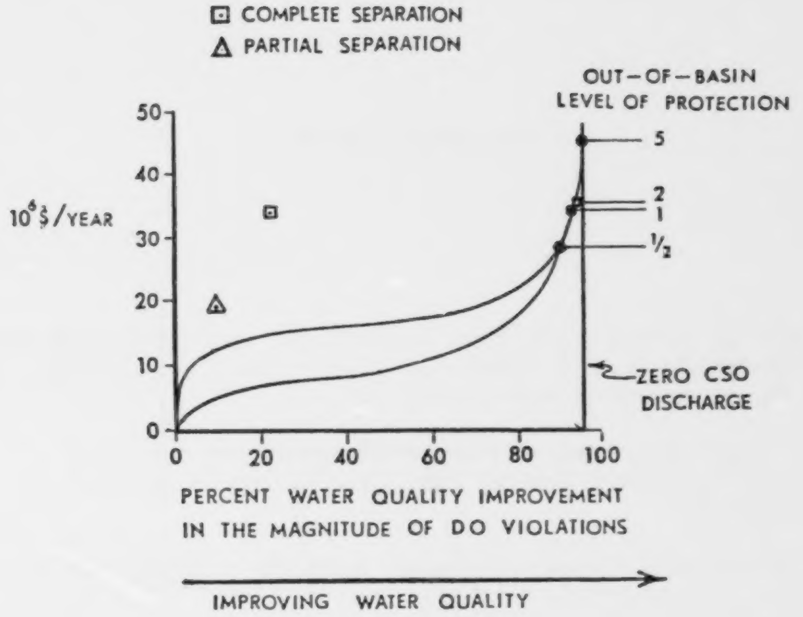


Figure 15. Cost versus water quality improvement and the magnitude of DO violations for the 1941 extreme runoff year at St. Paul Avenue.

aration points are also included in Figure 16 as well as lines indicating a possible shape of the curve. The problem of insufficient data to generate the cost versus water quality improvement curve is also evident with the fecal coliform results. Despite these problems, it is evident in the figures that the "knee" of the curve would occur at a level of protection of one-half year or less for both the DO and fecal coliform results.

Conclusion

This paper has attempted to briefly summarize the receiving water investigations that have been undertaken in conjunction with the CSO problem in Milwaukee. To date, the selection of the final solution for this problem involves a combination of sewer separation and storage treatment of the overflows. The final level of protection of the storage facilities will be dependent upon litigation presently being reviewed by the U.S. Supreme Court. If this litigation is overthrown, then the storage facilities would be designed for a two year level of protection.



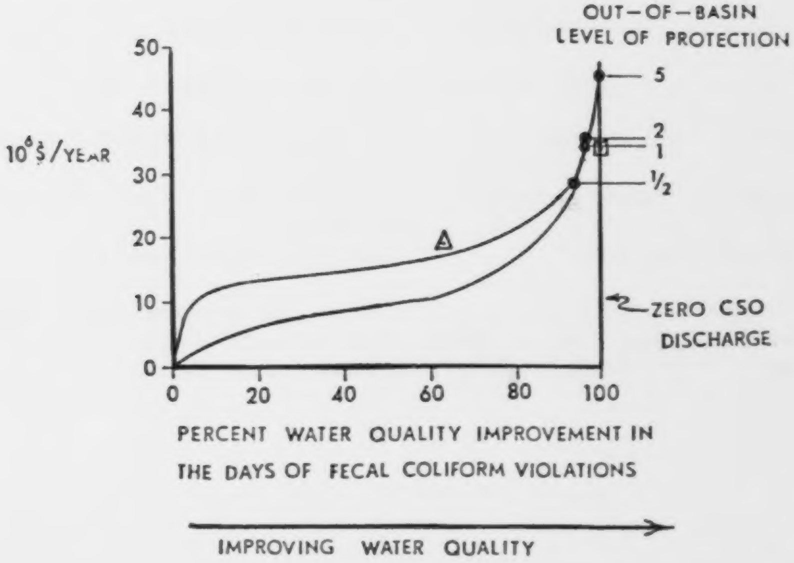


Figure 16. Cost versus water quality improvement for the 1954 extreme runoff year at St. Paul Avenue.

ACKNOWLEDGEMENT

This paper is based upon the results of two separate projects related to Milwaukee's Combined Sewer Overflow Study. The project final reports are entitled;

> "Verification of the Water Quality Impacts of Combined Sewer Overflow Using Real Time Data"

for the Storm and Combined Sewer Section - USEPA Edison, N.J.; and

> "Water Quality Analysis of the Milwaukee River to Meet PRM 75-34 (PG 61) Requirements"

for the Milwaukee Metropolitan Sewerage District.

Both of these project reports were co-authored by

Thomas L. Meinholz	presently of EcolSciences, inc. Milwaukee
William A. Kreutzberger	Rexnord Env. Res. Center, Milwaukee
Martin E. Harper	Harper-Owes, Seattle

The principal author would also like to acknowledge Messrs Richard Field and John English - USEPA and Messrs James Meinholz and James Ibach - Milwaukee Metropolitan Sewerage District.

URBAN STORMWATER IMPACTS ON RECEIVING STREAMS IN NORTH CAROLINA

by

E. Ryland Brown, Env. Scientist
Ross S. Green, Env. Biologist
N.C. Department of Natural Resources
and Community Development
Division of Environmental Management
Raleigh, North Carolina 27611

ABSTRACT

Studies were conducted in North Carolina by the Division of Environmental Management as part of the statewide 208 program to determine the extent of water quality degradation in streams receiving urban stormwater runoff. Selected streams were monitored in three of the larger urban areas within the state; Asheville, Raleigh, and Winston-Salem. The monitoring program involved physical/chemical sampling under both high (storm) and low streamflow conditions. Parameters sampled include pH, temperature, D.O., BOD_5, COD, fecal coliform, ammonia-nitrogen, total Kjeldahl nitrogen, nitrite plus nitrate nitrogen, total phosphorous, iron, mercury, lead, zinc, copper, cadmium, chromium, nickel and solids. Biological sampling of aquatic benthic macroinvertebrates was conducted concurrently to further document water quality conditions.

In Asheville, a stream draining a mixed land use urban area was studied. In the Raleigh area, three streams were monitored, including one that drains a highly impervious watershed comprised primarily of a shopping center. The most intensive sampling was conducted in the Winston-Salem area. In this locality, streams draining both residential and Central Business District watersheds were monitored to investigate water quality characteristics associated with these different land use types. For comparison purposes, control stations upstream of urban inputs were also established and monitored.

All of the urban streams were found to exhibit extensive water quality problems. Physical/chemical sampling consistently revealed high pollutant concentrations for several parameters under high flow conditions. Notable problematic parameters included suspended solids, lead, and some nutrients. Several parameters were found to frequently be present in high concentrations under low flow conditions also. Variations in pollutant concentrations from

the different land use types were also observed.

The Division of Environmental Management's biological monitoring group has developed an index of the macroinvertebrate organisms found in North Carolina streams. As part of this index, each taxa is rated according to its ability to withstand pollutional stressed conditions. All of the urban streams monitored were found to be extensively biologically degraded. Populations of Diptera and Oligochaeta, pollution-tolerant organisms, averaged over 90% of the existing fauna. In contrast, control station population percentages of these organisms were a maximum of 15% of the fauna in the mountain stream, and 35% in Piedmont streams. Many intolerant groups were not just reduced in numbers in the urban streams; they were usually completely absent.

These studies have shown that, under present conditions, almost all urban streams will be unable to meet the 1983 water quality goals. Improvement of water quality conditions in these streams is dependent upon the development of programs to effectively manage the entry of pollutants from urban land surfaces and unrecorded or intermittent point sources.

INTRODUCTION

Federal legislation enacted during the early 1970's provided the impetus for controlling the entry of pollutants from municipal and industrial discharges to the watercourses of the country. Since then, many advances have been made in the treatment of these discharges. These advances have greatly reduced the loads of various types of pollutants entering water bodies from these point sources.

In conjunction with the national commitment to limit point source discharges came the realization that significant inputs of pollutants were also contributed by non-point sources. Public Law 92-500, Section 208 provided the mechanism for addressing these non-point pollution sources. In North Carolina, one of the non-point pollution sources investigated was urban stormwater runoff. As part of this work, several streams were monitored in three of the larger urban areas within the State. The purpose of these studies was to determine the impacts associated with urban runoff and the management programs needed to address any observed impacts.

The objective of this paper is to present the results of monitoring activities conducted on these streams. A methodology is presented that utilizes the monitoring of aquatic benthic macroinvertebrate communities to complement traditional physical/chemical sampling techniques. The results of both sampling approaches are used to define the water quality status of the streams.

THE ENVIRONMENTAL SETTING

The southeast United States, extending from Maryland to Alabama, is characterized by three distinct physiographic regions: the Appalachian mountains, the Piedmont Plateau, and the Coastal Plain. Urbanization is greatest in the Piedmont, due largely to the abundant supply of water that has attracted industry. This urbanization is characterized by numerous medium size cities (populations of 30,000 to 150,000) located primarily along the major watercourses.

North Carolina is primarily a rural state. In 1970, only 23.9 percent of the State's population lived in designated urbanized areas, in contrast to 58.3 percent nationwide (1). The Piedmont is the most populated region, and the trend towards urbanization in this part of the State is increasing. It has been estimated that in 1980 the Piedmont will contain 52 percent of the State's population while 38 percent will live in the Coastal Plain and 10 percent in the mountains (2). The locations of the physiographic regions and larger cities in North Carolina are shown in Figure 1.

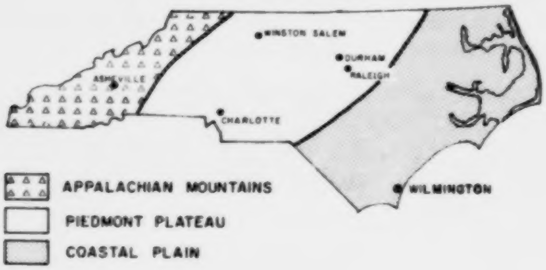


Figure 1. Location of the Physiographic Regions and Larger Cities in North Carolina.

The majority of the municipalities in the State have separate sewer systems for sewage and stormwater. The older Central Business Districts are usually served by underground storm sewers, with construction ranging from brick in the older areas to concrete in the newer lines. These inner-city areas are normally surrounded by well-established old residential areas with open channeled streams. Beyond this are generally rapidly developing suburban areas.

URBAN STREAM SAMPLING TECHNIQUES

Characterization of the magnitude and importance of urban stormwater runoff is a difficult task. Sources of pollutants are diverse and hard to characterize. In most cases, the pollutants entering streams via urban drainage originate from a mixture of sources: washoff from urban land surfaces; atmospheric fallout and washout; garbage deposited in or near streams; polluted groundwater; leaking or broken sanitary sewers; cross connections between sanitary and storm sewers; sanitary sewer overflow; small, unpermitted or intermittent point sources; or benthic resuspension of material settled out in pipes or stream pools. All of these sources must be considered as to their relative importance when determining a management program for a specific area.

Urban stream sampling is likewise not an easy task. Periodic grab sampling under low and high flow conditions can provide considerable information concerning the range of pollutant concentrations that are found in the streams. Concentrations can then be compared to established water quality standards to provide a mechanism for determining relative water quality conditions. Automatic sampling equipment, when available, can give more complete data records for impact assessment. However, this approach can be very expensive when applied on a large scale, and instrument malfunctions can result in incomplete data records upon which to base total pollutant loadings.

The costs of chemical analyses often limit the number of samples and parameters that can be analyzed in urban stream monitoring programs. Therefore, certain pollutant parameters or possible high concentration pollutant slugs that can cause aquatic biological degradation may not be detected in normal monitoring efforts.

Sampling for aquatic benthic macroinvertebrate organisms can provide a valuable mechanism for quantifying the extent of stream degradation. When used in conjunction with physical/chemical sampling techniques, a complete picture of problematic pollutants and resultant in-stream effects may be acquired.

STUDY LOCATIONS

Urban streams in the cities of Asheville, Raleigh, and Winston-Salem were sampled between fall, 1977 and early 1979. Names, predominant land use, location and drainage area of these streams are included in Table 1.

TABLE 1. CHARACTERISTICS OF STREAMS STUDIED

Stream	Drainage Area (mi^2)	Predominant Land Use	City
Sweeten Creek	5.2	Commercial/Industrial	Asheville
Pigeonhouse Branch	2.4	CBD/Commercial	Raleigh
Tar Branch	0.4	CBD	Winston-Salem
Salem Creek	31.7	Mixed Urban	Winston-Salem
Silas Creek Trib. 1	0.5	Residential	Winston-Salem
Silas Creek Trib. 2	0.4	Residential	Winston-Salem

Additionally, study stations were located upstream of the urbanized areas along Sweeten Creek and Salem Creek. Another Piedmont province station was located on Fourmile Creek, a minimally polluted stream. These stations served as control localities for comparisons of water quality characteristics between areas receiving and not receiving urban drainage.

Sweeten Creek receives considerable urban drainage from the city of Asheville (population 60,000). The stream runs parallel to many businesses and small industries. Recent studies have also indicated that broken sewer lines contribute drainage to the creek on an intermittent basis. The monitoring station is located about one stream mile into the commercial section. The drainage area is about 5.2 square miles at this point, and no significant point source dischargers are known to exist. The control station upstream of the urbanized area is located at the headwaters of Sweeten Creek, approximately ½ mile upstream of Busbee Reservoir.

Pigeonhouse Branch, an urban stream in Raleigh (population 150,000), drains a watershed consisting of a large shopping center, high density residential areas, commercial areas and small industries. Two monitoring stations were located on the stream. The first station is located at a point where the drainage area is only about 100 acres, and is primarily made up of the shopping center. Approximately 65 percent of the watershed is impervious due to the extensive parking lots, street surfaces, and rooftops. Resultant high volume and high velocity stormwater runoff has caused extensive downstream streambank degradation. The second station is located near West Street. At this point, the watershed is approximately 2.4 square miles in size and the stream has received drainage from the various residential and commercial areas in addition to the shopping center.

Tar Branch drains the Central Business District (CBD) of Winston-Salem (population 150,000). The watershed is highly impervious and is approximately 0.4 square miles in size. Much of the stream runs through pipes and culverts. It becomes open channel flow approximately 25 feet upstream of the established sampling station. The stream is approximately two meters wide and averages seven centimeters deep there.

The station located on Salem Creek downstream of Winston-Salem is near Silas Creek Parkway. At this point, the stream has received drainage from the Central Business District via Tar Branch plus drainage from various residential, commercial and industrial land use areas. The control station on Salem Creek is located approximately 1 mile downstream of Salem Lake. The stream is four meters wide and has an average depth of 30 centimeters at this location.

The two Silas Creek tributaries in Winston-Salem drain residential areas. Tributary 1 drains a watershed consisting mainly of medium density, high income housing. The second tributary drains an area that is mainly medium density, middle income housing. The watersheds were chosen because they are both well established, have little ongoing construction, and well represent typical residential areas in the larger cities within the state.

PHYSICAL/CHEMICAL SAMPLING

Physical/chemical sampling was conducted on a nonperiodic basis at the monitoring stations beginning in spring, 1977 and running through 1978. Samples were collected by hand using a DH-48 depth-integrated sampler. Monitoring was conducted under both normal (low flow) conditions and storm (high flow) conditions. Parameters sampled include pH, temperature, D.O., BOD_5, COD, fecal coliform, ammonia-nitrogen, total Kjeldahl nitrogen, nitrite plus nitrate nitrogen, total phosphorous, iron, mercury, lead, zinc, copper, cadmium, chromium, nickel, and solids.

Data collected were examined based on a comparison of the observed pollutant concentrations to proposed North Carolina water quality standards. Those samples having concentrations in excess of the proposed standards were termed "violations". This is a useful expression, since it is envisioned that when non-point source compliance monitoring programs are established, it will be necessary to express allowable limits for various pollutants on a probablistic or frequency basis (3). It would be impractical if not impossible to comply with pollutant limitations 100 percent of the time.

The data collected demonstrated some obvious urban-related water quality problems. High concentrations of several parameters and "violations" of proposed water quality standards were found in all of the urban streams under storm flow conditions. Additionally, monitoring showed that water quality problems also exist during low flow conditions. Several of the streams were reclassified in 1977 to support fish and wildlife. The streams had been previously classified only for urban and industrial conveyance usage. In many cases, the degradation of water is to the point that it cannot support fish life, and the poor quality exists during normal as well as storm flow conditions.

Distinct differences in pollutant concentrations were observed for several parameters in samples from the CBD and residential areas in Winston-Salem. Concentrations of most pollutants were found to consistently be higher in the CBD samples. This observation is consistent with other investigations of urban streams (4,5,6), and is important in determining ways of abating urban water pollution problems.

Nutrient Concentrations: Nutrient parameters monitored included NH_3, TKN, $NO_2 + NO_3$, and TP. The most intense sampling for these parameters was conducted in the Winston-Salem area. Concentrations of all these parameters were generally found to be considerably higher in the CBD stream than in the residential streams under low flow conditions. Concentrations at the control station downstream of Salem Lake were usually even lower. Table 2 shows the average concentrations of the nutrient parameters for each of these stream stations.

Concentrations of NH_3 and TKN were generally quite low in the low flow samples collected in the residential area streams, and were similar to the values recorded at the Salem-Creek control station. In contrast, concentrations observed at the CBD station were considerably higher for these parameters. Concentrations fluctuated over a much wider range in the CBD samples also. Values recorded at the downstream Salem Creek Station were intermediate between the residential and CBD station values, possibly reflecting the mixed land use drainage received by the stream at this point.

Concentrations of $NO_2 + NO_3$ were found to be even more visibly different between the CBD, residential, and control stations under low flow conditions. Mean quarterly concentrations were found to be several times higher in the CBD watershed for all quarters. Control station concentrations were consistently lower than those in both the CBD and residential samples. Average values observed at the downstream Salem Creek station were between the CBD and residential samples, again possibly reflecting the mixed land use drainage to the stream.

TABLE 2. MEAN LOW-FLOW NUTRIENT CONCENTRATIONS
IN WINSTON-SALEM STREAMS

Stream	Watershed Type	Mean Low-Flow Nutrient Concentration (mg/1)*			
		NH_3	NO_2+NO_3	TKN	TP
Salem Creek Control	Rural	.08	.27	.29	.07
Silas Creek Trib. 1	Residential	.04	1.00	.19	.16
Silas Creek Trib. 2	Residential	.05	.52	.18	.20
Tar Branch	CBD	.56	4.48	1.56	.24
Salem Creek Downstream	Mixed Urban	.20	.89	.45	.09

*Average of 20 samples taken during 1978.

Average total phosphorous concentrations did not vary greatly between samples collected at the different stations. Average values were consistently low, ranging from 0.07 mg/l at the control station to 0.24 mg/l at the CBD station.

Concentrations of each of these parameters generally tended to be greater during high flow conditions, as compared to low flow conditions, although this was not always the case. These elevated concentrations, when they occurred, were not as pronounced as those of some of the heavy metal parameters. No visible seasonal concentration differences were evident in this study.

Heavy Metal Concentrations: Table 3 shows the monitoring results for the heavy metal parameters of cadmium, iron, lead, mercury, and nickel for data collected over the first three quarters of 1978 in the Winston-Salem area. It is important to note that all metals did not show the same relationships. For instance, under low flow conditions, iron "violations" were 8 percent in the residential area, and 13 percent in the CBD. At the downstream station which receives flow from both of these land use types, "violations" occurred in 25 percent of the samples. Average iron concentrations appeared higher in the CBD watershed than in the residential areas. However, the upstream control station was found to have the highest "percent violation" for iron. This may reflect the natural high iron content of the soils in this area. Mercury concentrations were similar throughout the different land uses, including the upstream control area. "Violations" averaged 45 percent. Under low flow conditions, lead "violations" appeared highest in the residential areas; no "violations"were found in the upstream control samples. Storm flow samples were found to have much higher lead concentrations than low flow samples in the CBD. "Violations" occurred 100 percent of the time in samples collected during storm events.

Zinc concentrations appeared higher in the CBD than in the residential areas. Percent "violations" are not expressed in this work since proposed North Carolina standards are based on a 96-hour LC50 figure rather than in terms of concentration for this parameter. There were no concentrations of cadmium and chromium, and very few concentrations of nickel high enough to constitute "violations" of the proposed standards under either low or high flow conditions.

TABLE 3. HEAVY METALS "VIOLATIONS"

Standards for Five Toxics Investigated

.0211 (a)(4) General Toxic Standards (7)

Cadmium: not greater than 4.0 ug/l for non-trout waters
Iron: not greater than 1.0 mg/l for non-trout waters
Lead: not greater than 0.03 mg/l for non-trout waters
Mercury: not greater than 0.05 ug/l for non-trout waters
Nickel: not greater than 0.1 mg/l for non-trout waters

"Percent Violations"

	Low Flow Avg.	Storm Flow Avg.
Cadmium	0% for all samples taken	
Iron		
Upstream (Salem Creek Control)	37%	
Residential (Silas Creek Tribs)	8%	
CBD (Tar Branch)	13%	92%
Urban (Salem Creek Downstream)	25%	
Avg. (All Stations except Control)	15%	
Lead*		
Upstream	0%	
Residential	20%	
CBD	8%	100%
Urban	10%	
Avg.	13%	
Mercury		
Upstream	40%	
Residential	45%	
CBD	50%	13%
Urban	40%	
Avg.	45%	
Nickel		
Upstream	0%	
Residential	3%	
CBD	8%	9%
Urban	0% for all samples taken	
Avg.	4%	

*The proposed standard for lead is below the laboratory detection limit of

100 ug/l. The reported "violations" were samples with concentrations of 100 ug/l or greater. Some of the samples with concentrations less than 100 ug/l may actually also be greater than the proposed 30 ug/l standard.

Mercury concentrations appeared to be higher under low flow conditions than under high flow conditions. Stations in the CBD evidenced proposed standard "violations" 50 percent of the time under low flow conditions; high flow "violations" there were observed 13 percent of the time. Concentrations of lead, iron, and zinc, however, were found to be many times higher under wet than under dry flow conditions (Figure 2). Iron, for example, was found in concentrations in excess of the proposed standards 92 percent of the time under high flow conditions in the CBD; dry flow "violations" averaged 13 percent.

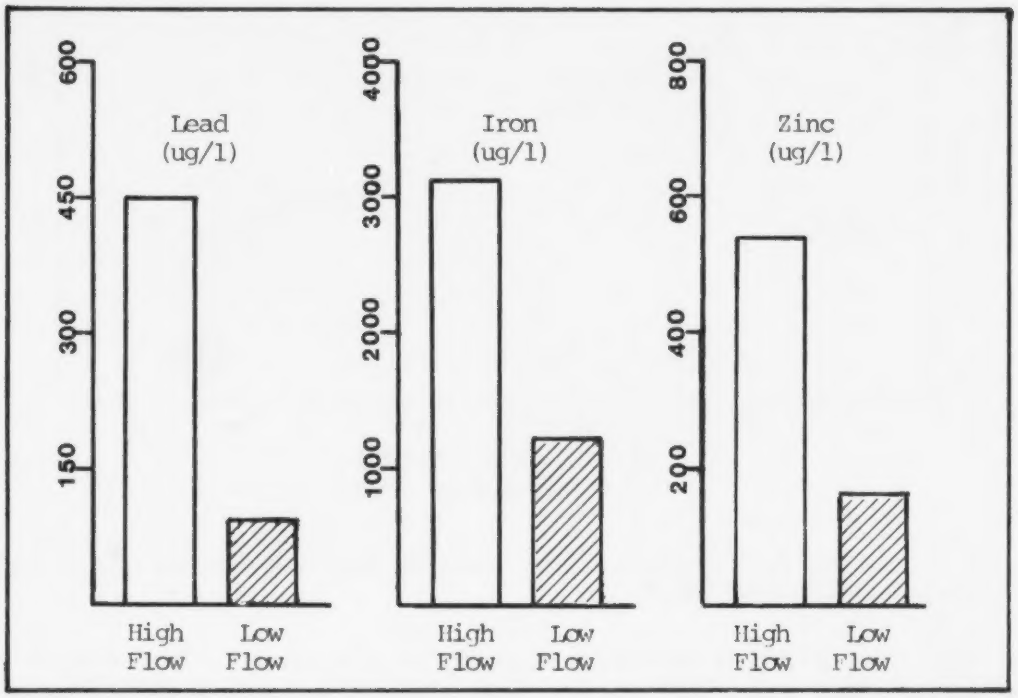


Figure 2. Average Concentrations of Heavy Metals in Streams Draining the CBD Under Different Flow Conditions.

Samples taken from Pigeonhouse Branch in Raleigh were also found to frequently have quite high concentrations of certain heavy metals under both high and low flow conditions. Lead concentrations were found to be extremely high under storm flow conditions. Figure 3 shows lead concentrations observed during the storm of May 23, 1977. In all cases, concentrations of lead were well above 30 ug/l proposed standard. From the graph, it appears that this watershed exhibits a "first flush" of lead during a storm event. The highest concentration of lead observed during the study was 6000 ug/l.

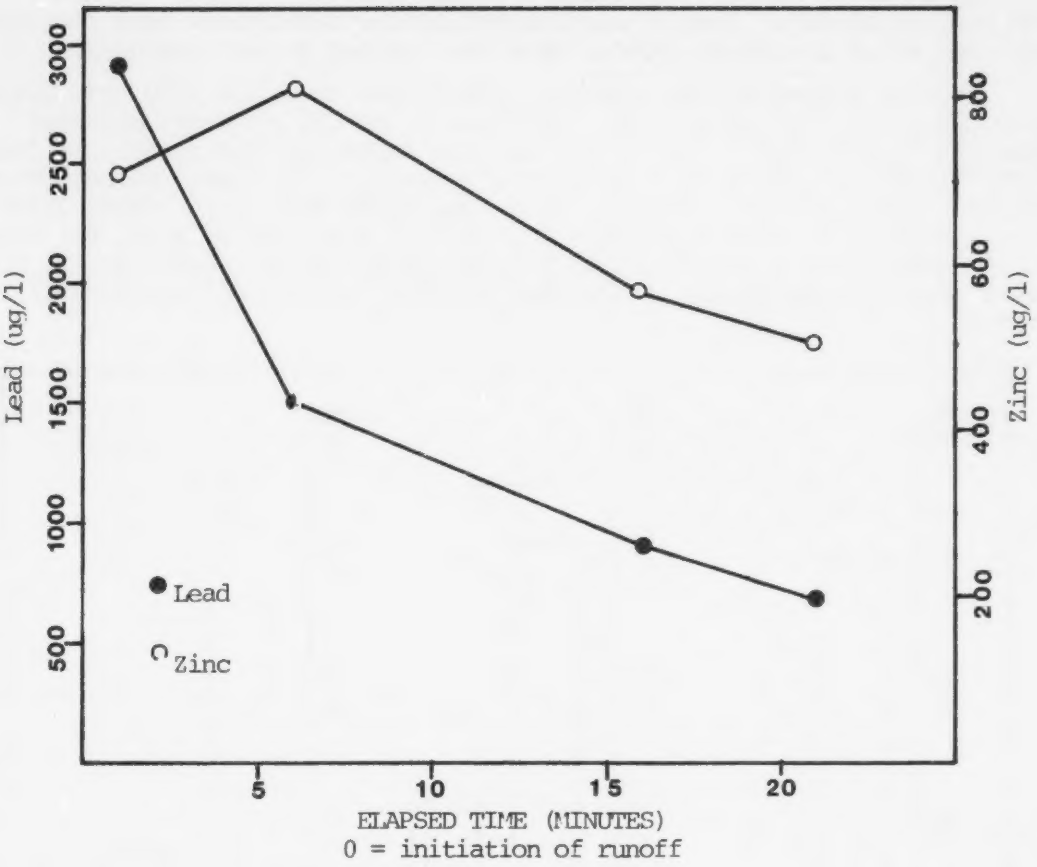


Figure 3. Lead and Zinc Concentrations Observed During the Storm of May 23, 1977 (Pigeonhouse Branch Watershed).

Zinc concentrations were also found to be relatively high in several of the samples, both under high and low flow conditions. The concentrations of zinc found during the storm of May 23, 1977 are also shown in Figure 3. The peak concentration of zinc was observed later in the storm than that for lead, i.e. this pollutant did not appear to be washed off in a first flush manner as pronounced as that of lead. Other samples collected during the study registered concentrations as high as 2100 ug/l.

One factor leading to the extremely large concentrations of these metals observed during runoff events may be the extensive impervious areas within the watershed and the associated potential for pollutant buildup and washoff. A secondary factor may be the resuspension of pollutants that have settled out in pipes or in pools of open channels during previous periods of dry weather.

Oxygen-Demanding Material Concentrations: Both five day biochemical oxygen demand (BOD_5), and chemical oxygen demand (COD) were monitored during the course of this study. Quite wide concentration ranges were evident for both of these parameters at both the CBD and residential area stations in Winston-Salem under low flow conditions. BOD_5 values were consistently greater in the CBD samples than those from the residential areas. BOD_5 values reached a maximum of 18.6 mg/l in the CBD; the highest residential area value was 5.94 mg/l. Figure 4 shows a plot of BOD_5 concentrations found at one of the CBD stations and the upstream control station below Salem Lake under low flow conditions. As can be seen, BOD_5 concentrations were always greater at the urban stream station, often being many times higher. This clearly shows the type of conditions that can be associated with urban-affected streams.

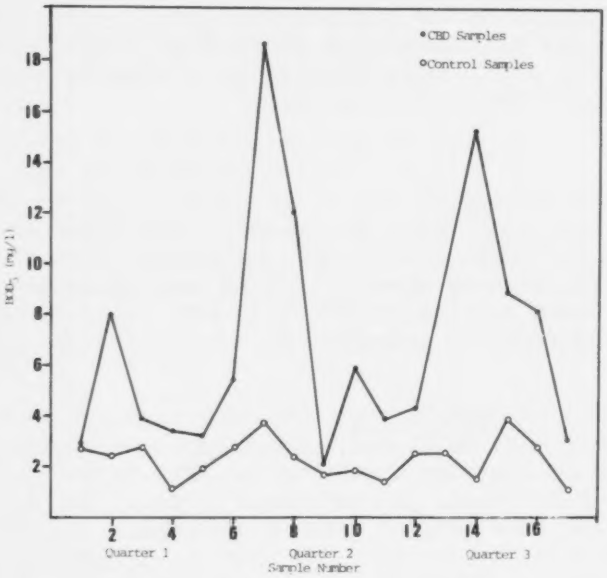


Figure 4. BOD_5 Concentrations observed at Winston-Salem CBD and Control Stations.

BOD_5 concentrations were found to be generally much higher under storm flow conditions than under low flow conditions in the CBD watershed. Storm flow concentrations as high as 27.9 mg/l were observed. The average BOD_5 value for six samples taken at the CBD station during the storm of March 8, 1978 was 24.1 mg/l. In comparison, the average of 17 low flow samples taken at this station during 1978 was 7.0 mg/l.

COD concentrations were accordingly higher in the CBD area than in the residential areas of Winston-Salem also. Mean quarterly COD sample concentrations were greater at the CBD station in all cases. Additionally, the range of COD values was much wider at the CBD station. However, there was not as visible a difference in concentration ranges between the two land use types as there was for BOD_5 concentrations. The maximum COD level observed in the CBD watershed was 49 mg/l. The high value observed in the residential area was 39 mg/l; most values were considerably lower.

High flow COD concentrations were found to be considerably higher than low flow values in the CBD watershed. A maximum COD of 210 mg/l was found at the CBD station during the storm of March 8, 1978. The average of six samples taken during this storm was 130 mg/l COD. Six samples taken at the same station during the storm of March 14, 1978 had an average COD of 73 mg/l. In comparison, 20 low flow samples taken at the station during 1978 averaged 21 mg/l.

In the Raleigh area, even larger concentrations of oxygen-demanding materials were found to enter the receiving stream under storm flow conditions. Both BOD and COD levels were quite high in the majority of the storm flow samples. BOD_5 levels were found to be as high as 89 mg/l compared to the 0-4 mg/l range found in most streams.

Figure 5 shows the BOD_5 concentrations observed during the storm of August 1, 1977. It can be seen that oxygen-demanding materials were washed off in the largest concentrations in the early stages of the storm in accordance with the first flush phenomenon. Peak concentrations were found between five and ten minutes after the initiation of runoff. The corresponding COD concentrations observed during this storm are also shown in Figure 5. COD values as high as 520 mg/l were observed. The wide COD/BOD_5 ratios suggest the possible presence of toxic materials, inhibiting biochemical oxidation.

Fecal Coliform Concentrations: Concentrations of fecal coliform microorganisms were also monitored. Concentrations varied greatly over the study period, but on numerous occasions the 1000 microorganisms/100 ml proposed standard was exceeded. A wide range of fecal coliform concentrations was observed in low flow samples taken from streams draining the two different land uses in Winston-Salem. No readily visible difference in mean concentrations could be detected between the two areas. Maximum concentrations observed were 270,000 mo/100 ml in the CBD area and 190,000 mo/100 ml in the residential areas. Concentrations observed during storm flow conditions were also high on several occasions. The data serves to show that fecal coliform contamination from some sources within the urban area is occurring during both wet and dry periods. This observation is borne out by the fact that no samples at the upstream control station had concentrations large enough to exceed the proposed samples. At this station, the mean low flow concentration was 77 mo/100 ml, and the maximum concentration was 340 mo/100 ml.

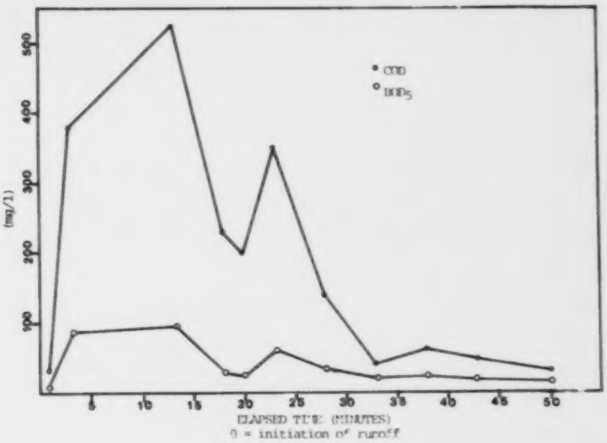


Figure 5. BOD_5 and COD Concentrations Observed During the Storm of August 1, 1977 (Pigeonhouse Branch Watershed).

Low flow fecal coliform concentrations averaged 2600 in Sweeten Creek in work conducted in conjunction with the Land-of-Sky Regional Council of Governments (8). In comparison, stormflow samples contained concentrations averaged 8,233 mo/100 ml.

Suspended Solids: Solids were found to be present in large concentrations during periods of storm flow. Samples taken during storm flow periods exceeded the proposed 80 mg/l standard in almost all cases. Figure 8 shows the total solids and total suspended solids observed in Pigeonhouse Branch during the storm of August 1, 1977. An obvious first flush response was found to occur for this parameter, with total suspended solids values as high as 518 mg/l early in the storm. An interesting feature observed for the storm was that relatively large percentage of the solids in samples taken early in the storm were in the dissolved state; later samples were quite consistent in having solids mainly in the suspended state.

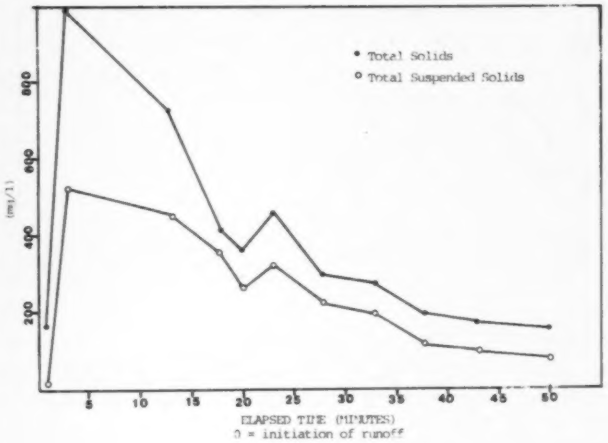


Figure 6. Solids Concentrations Observed During the Storm of August 1, 1977 (Pigeonhouse Branch).

BIOLOGICAL SAMPLING

The biological monitoring group has devised a system of analysis through which biological data can be evaluated to determine the degree of environmental stress from nonpoint source pollutants. The symptoms of nonpoint source pollution are reflected as changes in the biotic components of the environment. This shift in aquatic community structure (especially the benthic macroinvertebrates) is used as a means of measuring environmental impact. Macroinvertebrates are useful biological monitors because they are found in all aquatic habitats. Benthic invertebrates are less mobile than many other groups of organisms and are of a size which makes them easily collectable. Biological sampling is a useful supplement to physical and chemical studies because chemical studies cannot detect possible fluctuations in water quality between sampling periods. Short-term critical events may often be missed. The biota, however, reflect both long and short term conditions. Since most species in a macroinvertebrate community have life cycles of a year or more, the effects of a short term pollution will generally be revealed in the next biological sample. Chemical sampling for all potential pollutants can fail to evaluate possible synergistic and antagonistic effects. Macroinvertebrate sampling can integrate all forms of environmental stress and summarize their cumulative effects.

Nonpoint source pollutants can degrade water quality in a number of ways. The various effects of these pollutants on benthic communities has been summarized by Hocutt in Figure 7 (9). In clean, unstressed aquatic communities both density (N) and species richness (S) will have high numerical values (quadrant II). Toxic stress (including pesticides and metals) will reduce both S and N (quadrant III). Nutrient enrichment tends to increase the number of organisms (N) while selectively favoring a few types of organisms tolerant to this form of pollution (quadrant I). Moderate sediment stress is characterized by a decrease in density (due to habitat destruction) with little effect on species richness (quadrant IV).

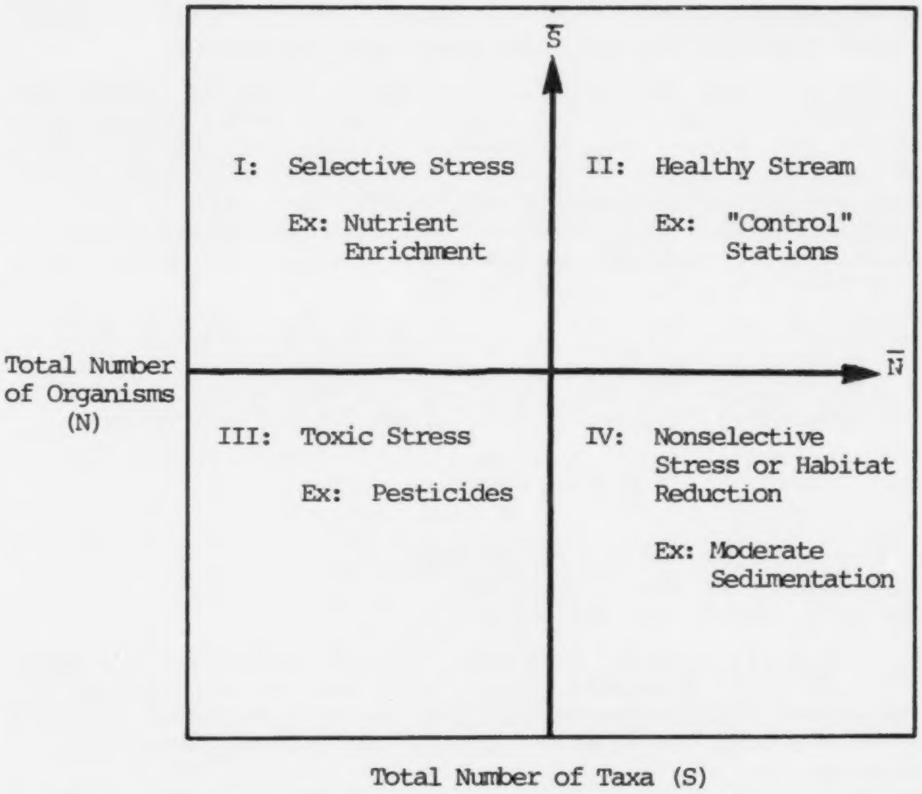


Figure 7. General Effects of Pollutants on Aquatic Macroinvertebrates.

Toxic pollutants have the most dramatic effect on stream benthos. Urban runoff often has associated heavy metals and pesticides, both extremely toxic to most stream benthos. Sediment along with nutrients in the form of sewage are also degrading components of urban runoff.

BIOLOGICAL SAMPLING PROCEDURES

Samples were collected using the "kick" method. A net is placed in the stream, while an upstream area of one square meter is physically disrupted. This method was chosen because it gives consistent results over a variety of substrates as compared with the surber and artificial substrates (10,11).

At each station a minimum of two samples were collected. Three samples in transect were taken on the larger streams. Samples were preserved in the field with ethanol. Physical parameters including DO, temperature, pH, and flow measurements were also taken in the field. Physical charcteristics such as substrate composition, percent bank cover, and surrounding land use were estimated during each sampling period. Samples were returned to the lab where they are sorted and identified to the lowest taxonomic level possible. All data were then evaluated using a three level analysis system.

Level I: Single Number Summaries: Single number summaries include the Shannon-Weiner diversity index, the N.C. regional biotic index, percent reduction of density (N), and percent reduction in species richness (S). The Shannon-Weiner index is a value that represents the relationship between the species richness and the total number of each species in a sample. High diversity is characterized by many different taxa with few numbers within each taxa. Relatively few taxa denotes a lower diversity. Unstressed communities tend to have the highest diversity values.

Biotic Index values are obtained by assigning an index value to each species collected (ranging from 0-5). 0 is assigned for very intolerant organisms and 5 is assigned when high tolerance is characteristic. A biotic index value for each sample is then obtained by the following formula,

$$B.I. = \frac{a_i \, n_i}{N}$$

where; a_i is the index value for the i^{th} species,
n_i is the density for the i^{th} species,
N is the total density for the sample.

Level II: Species Density and Richness: Density and species richness data is tabulated for major taxonomic groups. This data is then used in comparison with control data to assess reduction and elimination of sensitive taxa. Changes or shifts in the aquatic community may help to reveal the nature of pollutants.

Level III: Summaries by Species: Data at the species level can often be used to deduce the nature of pollutants. Information on the pollution tolerance of any given taxa when taken into consideration with the distribution of that organism often parellels any chemical and physical changes in the environment.

Guidelines shown in Table 4 were used to evaluate Level I and II data.

TABLE 4. LEVEL I AND II ANALYSIS CRITERIA
IN RELATION TO CONTROLS (12)*

A. Reduction in Density (N)
   1. 30% : Unstressed
   2. 30% - 60% : Moderate Stress
   3. 60% : Severe Stress

B. Reduction in Species (or Taxa) Richness (S)
   1. 20% : Unstressed
   2. 20% - 50% : Moderate Stress
   3. 50% : Severe Stress

C. Diversity ** (from 13)
   1. 3 : Clean
   2. 2 - 3 : Moderate Stress
   3. 2 : Severe Stress

D. Biotic Index (from 14)
   1. 1.8 : Excellent
   2. 1.8 - 2.3 : Good
   3. 2.3 - 3.0 : Fair
   4. 3.0 - 3.8 : Poor
   5. 3.8 : Very Poor

* Control stations or standard control data sets.
** A low diversity value usually reflects stress, but the converse is not always true.

In some geographical regions of North Carolina, it is virtually impossible to find an unimpacted site upstream to use as a control. To alleviate the problem, it is necessary to use a control data set which has been developed for each of the three geographical regions of the state (Table 5). This data set serves in the absence of a suitable control in the study area. The control data set is also used to verify an upstream control site. The control data set concept is a relatively new procedure which has given consistent results.

TABLE 5. SELECTED DATA FROM UNSTRESSED STREAMS AND RIVERS

A. Average * Taxa Richness (by Group)

Group	Mountain Streams (4)	Mountain Rivers (4)	Piedmont Streams (3)
Ephemeroptera	9.8	14.8	7.9
Plecoptera	6.8	6.7	2.8
Trichoptera	7.9	9.1	3.4
Diptera	13.8	19.2	16.2
Coleoptera	3.1	3.4	4.3
Other	3.9	1.9	9.3
Total	45.3	55.1	43.9

B. Density by Group (% of Total Density)

Group	Mountain Streams (4)	Mountain Rivers (4)	Piedmont Streams (2)**
Ephemeroptera	43	47	23
Plecoptera	14	8	9
Trichoptera	18	26	13
Diptera	13	12	32
Coleoptera	7	6	20
Other	5	1	3
Total	100	100	100

*Data represent the expected values for a single collection date combining all replicates. Streams with 2 replicates - Rivers with 3 replicates.

**Highly variable.

BIOLOGICAL SAMPLING RESULTS

Level I Analysis: Data from each of the urban streams are compared with control data in Figures 10-13. Sixty to ninety-five percent reductions in density (N) indicating severe biotic stress levels according to established criteria (12) were found at all streams, with the exception of Sweeten Creek Station 2. The stream at this station receives high levels of organic pollutants, which are responsible for a 186% increase in density (15). These density reduction results are shown in Figure 8. Density varied drastically among sampling periods. High density populations found following the influx of organic pollutants dropped drastically soon afterwards, indicating possible toxic pollutants.

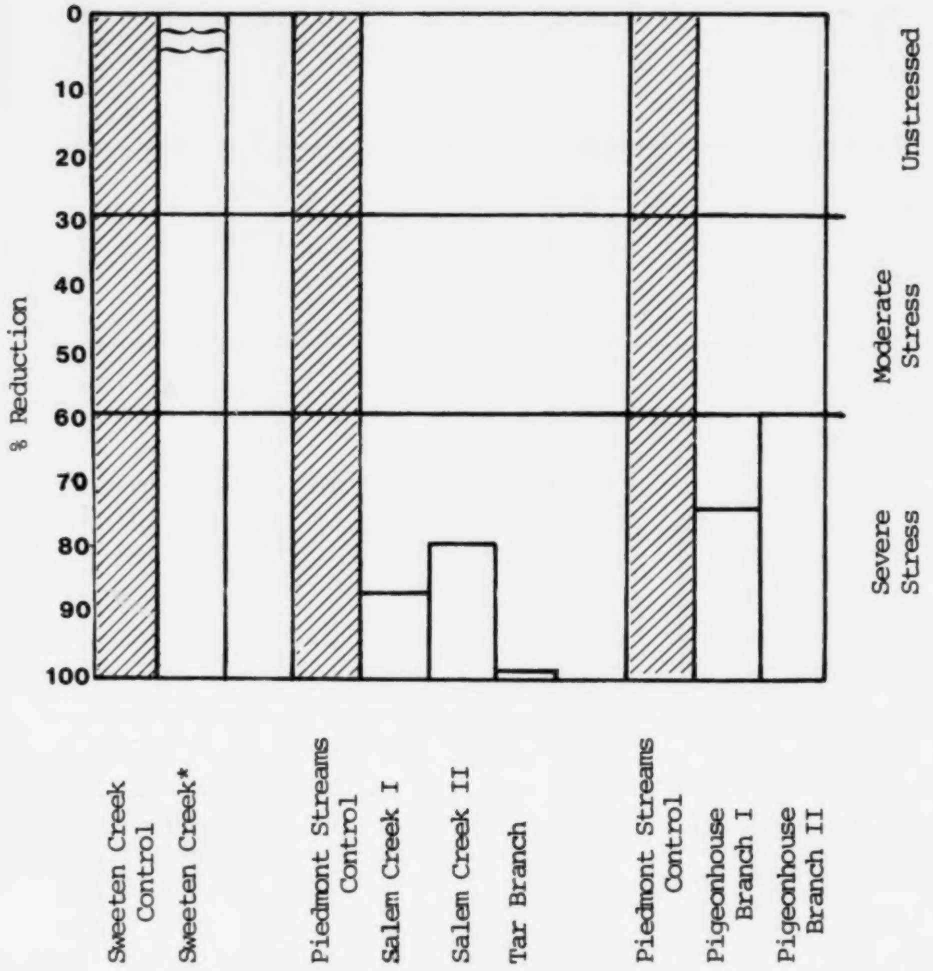


Figure 8. Analysis of Macroinvertebrate Communities. Level I Analysis: Density Percent Reduction (N).

Reduction in taxa richness (S) is shown in Figure 9, and illustrates severe stress at all urban stream sites. Severe stress is indicated when there is a 50 percent or greater reduction in taxa richness. All urban sites in this study were found to have reductions of 65-95 percent.

*denotes a 186% avg. increase in density which is highly variable (15).

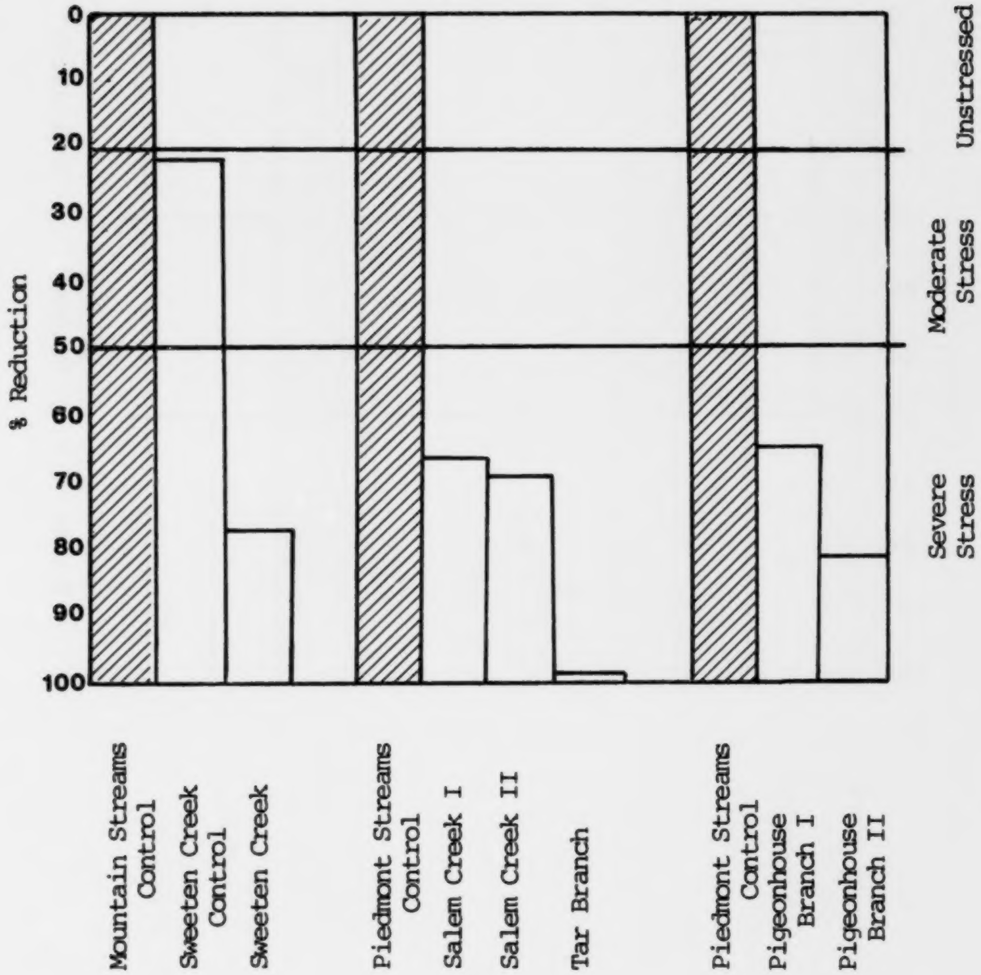


Figure 9. Analysis of Macroinvertebrate Communities. Level I Analysis: Taxa Richness Percent Reduction (S).

Diversity index values were calculated for all stations. Low diversity values were found at all urban stations, with the exception of Sweeten Creek Station 1, the control station (Figure 10). Criteria from Table 4 indicates severe stress at all other urban stations.

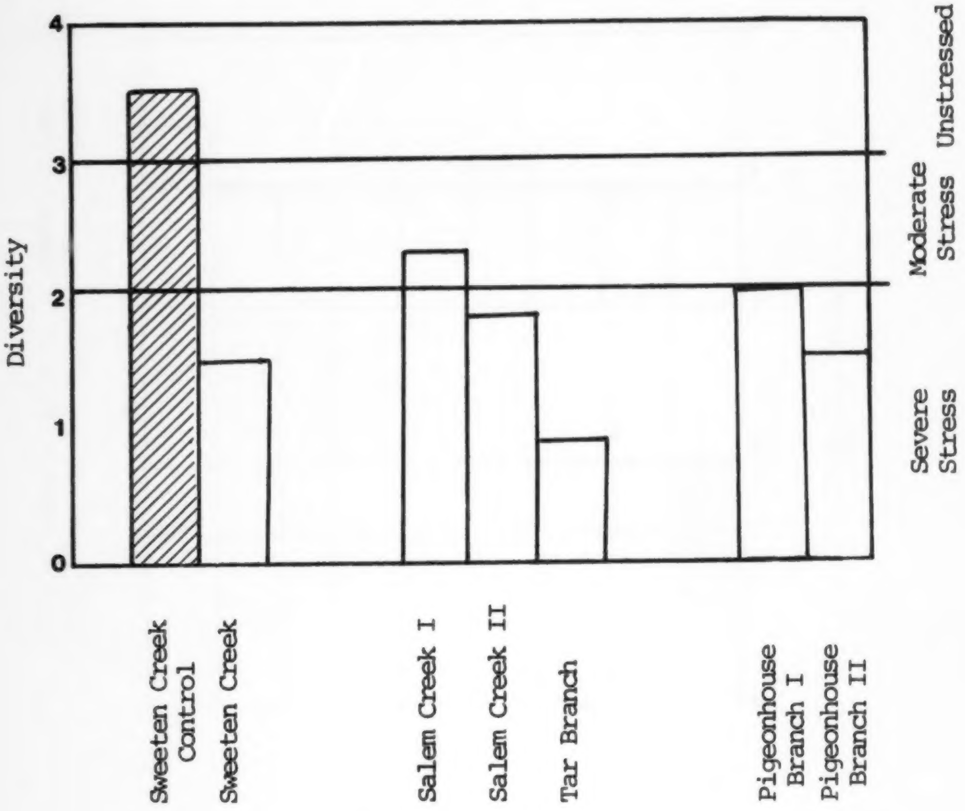


Figure 10. Analysis of Macroinvertebrate Communities. Level I Analysis: Average Diversity Index Values.

Average Biotic Index values were also calculated for all stations. All stations sampled, except the Sweeten Creek control site had Biotic Index values ranging from 3.4 to 4.3 (Figure 11). This range suggests poor to very poor water quality due to the dominance of pollution-tolerant taxa.

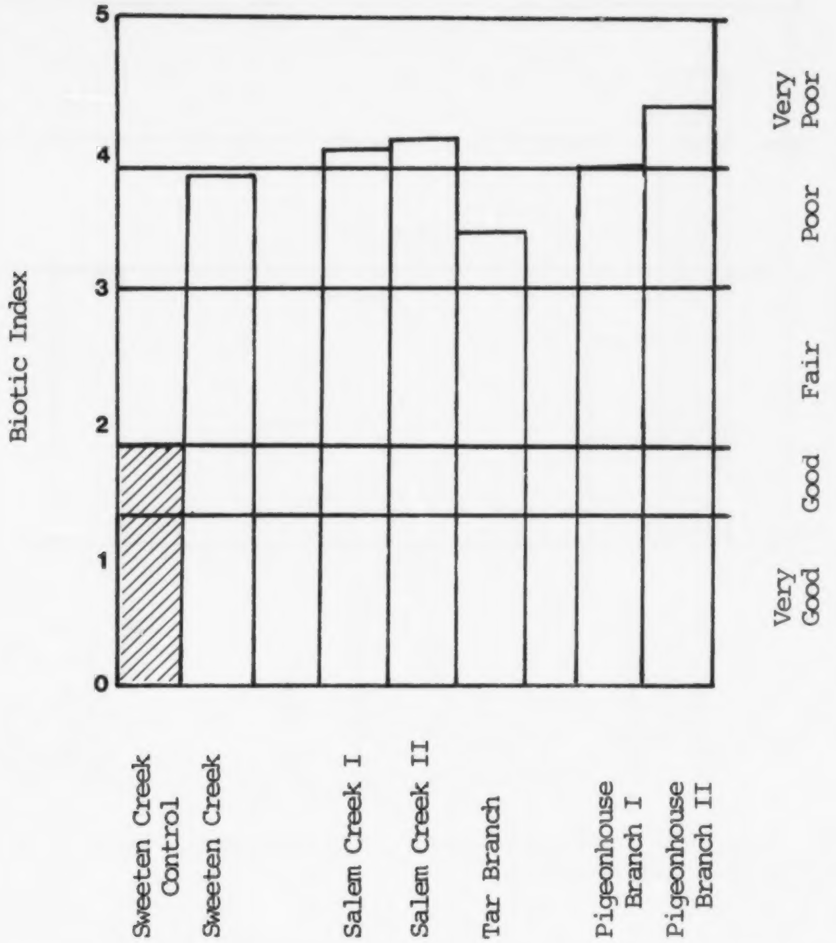


Figure 11. Analysis of Macroinvertebrate Communities. Level I Analysis: Biotic Index Values.

Level II Analysis: In analyzing the data from the three study areas, it is apparent that there is a sharp decrease in the number of sensitive macroinvertebrate groups present. Sweeten Creek station 1 (Control) is dominated by Ephyemeroptera, Plecoptera, Trichoptera, and Coleoptera; a typical clean water assembledge. However, station 2, the urban station, is dominated by Oligochaeta and Chironomidae, the most tolerant groups of organisms. Piedmont streams generally consist of a population in which Trichoptera, Plecoptera, and Coleoptera comprise approximately 40 percent of the sample at 65 percent of the density. These groups are not only reduced in all the urban study areas, but are virtually absent. Instead, 90 to 100 percent of the organisms are Oligochaeta and Chironomidae at these stations. Tables 6 and 7 show definite community shifts when observed urban station communities are compared to control station communities. Such extensive shifts in populations appear to be characteristic of all urban streams in North Carolina. There is not only a shift in the number of types of organisms found but also a density shift favoring the more tolerant taxa.

Group	Mountain Streams Control	Sweeten Creek Control	Sweeten Creek	Piedmont Streams Control	Salem Creek I	Salem Creek II	Tar Branch	Piedmont Streams Control	Pigeonhouse Branch I	Pigeonhouse Branch II
Ephemeroptera	9.8	4.8	--	7.9	0.8	--	0.2	7.9	0.3	--
Plecoptera	6.8	3.0	0.4	2.8	0.2	--	--	2.8	--	--
Trichoptera	7.9	5.8	0.4	3.4	0.4	0.2	0.2	3.4	--	0.3
Coleoptera	3.1	3.0	-	16.2	0.4	0.2	--	16.2	--	--
Diptera	13.8	9.2	4.6	4.3	8.2	6.2	2.0	4.3	11.3	5.3
Oligochaeta	--	1.0	2.6	--	2.6	4.2	1.6	--	2.5	2.5
Other	3.9	3.8	0.4	9.3	2.0	2.6	0.4	9.3	0.9	0.2
Total	45.3	30.6	8.4	43.9	14.6	13.4	4.4	43.9	15.0	8.3

Table 6. Analysis of Macroinvertebrate Communities. Level II Analysis: Taxa Richness by Group.

Group	Mountain Streams Control	Sweeten Creek Control	Sweeten Creek	Piedmont Streams Control	Salem Creek I	Salem Creek II	Tar Branch	Piedmont Streams Control	Pigeonhouse Branch I	Pigeonhouse Branch II
Ephemeroptera	43	17	--	23	3	--	3	23	--	--
Plecoptera	14	36	--	9	--	--	--	9	--	--
Trichoptera	18	27	--	13	--	1	4	13	--	--
Coleoptera	7	4	--	20	2	--	--	20	--	--
Diptera	13	13	10	32	40	21	63	32	9	62
Oligochaeta	--	1	89	--	42	77	23	--	90	37
Other	5	2	1	3	13	1	7	3	1	1
Total	100	100	100	100	100	100	100	100	100	100

Table 7. Analysis of Macroinvertebrate Communities. Level II Analysis: Taxa Density by Group (% of Total).

Level III Analysis: Level III data analyses support conclusions found in prior sections. Control urban station data from all three study areas were evaluated at the species level. A marked shift in community composition between the control and urban stations was evident. The control communities were found to be dominated by relatively intolerant species from several orders. These organisms are virtually absent at all urban impacted stations. It is important to also note that the species that do survive at the urban sites are very tolerant organisms to most forms of pollution. These organisms typically include the oligocheates *Limnodrilus spp*. and *Limnodrilus hoffmeisteri* and the tolerant Chironomidae (i.e. *Chironomus* sp. and *Polypedilum illinoense*). Table 8 illustrates the ten species that are characteristic of the urban streams studied. The presence of these organisms in urban streams as opposed to those found at the control stations is attributed to the extensive water quality pollution resulting from urban drainage.

TABLE 8. CHARACTERISTIC FAUNA OF URBAN STREAMS

TAXA	COMMENTS
Oligochaeta	
Limnodrilus hoffmeisteri	Common
L. udekamianus	Irregular distribution, usually associated with high organics
Ilyodrilustempletoni	A species often associated with *L. hoffmeisteri*
Nais sp. (Prob: *N. Communis*)	Highly sporadic distribution. High densities usually found only in spring, especially under low flow conditions.
Lumbriculidae	Common, but rarely with high density
Chironomidae	
Chironomus sp.	Rarely present in the high density expected for simple organic pollution.
Polypedilum illinoense	Common. The most tolerant species in the genus *Polypedilum*.
Tribelos sp.	Occasionally abundant
Cricotopus spp.	Common
C. sp 5/14	*C. tremulus* gr, heavy crenulations of mandible
C. sp 6	*C. tremulus* gr, crenulations largely absent
C. sp 1	*C. bicinctus* gr
Conchapelopia sp.	Common

CONCLUSIONS

1) Sources of pollutants entering urban streams are very complex and include urban washoff, atmospheric washout and fallout, ineffective sanitary sewers, and small, intermittent or unpermitted point source discharges. All of these sources must be considered as to their relative importance in the development of management programs that will effectively improve the quality of urban streams.

2) Physical/chemical sampling of urban streams has revelaed numerous water quality problems. Several pollutant parameters were found to be in excess of proposed water quality standards, and these high concentrations occasionally occurred under low flow as well as high flow conditions.

3) The collection of stream samples on a periodic basis and a comparison of the concentrations observed for selected pollutants versus adopted water quality standards may be a useful tool in expressing allowable nonpoint discharge limits in conjunction with nonpoint source permitting programs if they are instituted.

4) Residential and Central Business District land uses within urban areas have distinctly different pollutant loading characteristics, and require different management approaches. It is apparent that restoration of water quality in inner-city streams will require considerably more effort than needed for streams draining residential areas.

5) Pollutants settled out in pipes or stream pools under periods of low flow may be resuspended under subsequent storm flow conditions. Such benthic resuspension may result in highly polluted plug flows leading to severe downstream biological stress.

6) The sampling of benthic macroinvertebrate organisms can provide a valuable tool to be used in conjunction with established physical/chemical sampling methodologies for assessing the impacts of urban stormwater runoff.

7) All of the urban streams monitored exhibited extensive water quality degradation. Observations on other urban streams in North Carolina have shown similar water quality problems, and it is apparent that under present conditions, almost all urban streams will be unable to meet the established 1983 water quality goals.

REFERENCES

1. Hyman, D.N. Urbanization in North Carolina. Economics Information Report No. 38, North Carolina State University, Raleigh, North Carolina, 1974.
2. Clay, J.W., et al. North Carolina Atlas, Portrait of a Changing Southern State. University of North Carolina Press, Chapel Hill, North Carolina, 1975.
3. McPherson, M.B. Challenges in Urban Runoff Control. Paper presented at the 14th Annual Henry M. Shaw Lecture in Civil Engineering, Raleigh, North Carolina, March 13, 1977.

4. Grizzard, T.J., J.P. Hartigan, C.W. Randall, J.I. Kim, J.T. Smullen, and M. Dere Wianka. Assessing Runoff Pollution Loadings for 208 Planning Programs. In: Proceedings of the ASCE National Environmental Engineering Conference, Nashville, Tennessee, 1977.
5. Colston, N.V., Jr. Characterization and Treatment of Urban Land Runoff. EPA - 670/2 = 74 - 096, U.S. Environmental Protection Agency, Cincinnati, Ohio, 1974.
6. Triangle J Regional Council of Governments. Pollution Source Analysis. Research Triangle Park, North Carolina, 1976.
7. Division of Environmental Management, N.C. Department of Natural Resources and Community Development. Proposed Revisions to North Carolina's Water Quality Standards, Draft #6. Raleigh, North Carolina, March 1, 1978.
8. Land-of-Sky Regional Council of Governments. 208 Areawide Water Quality Management Plan, Asheville, North Carolina, 1978.
9. Hocutt, C.H. Assessment of a Stressed Macroinvertebrate Community. Water Resources Bulletin, 11 (4): 820-835, 1975.
10. Kinney, W.L., V.E. Pollard, and C.E. Hornig. Comparisons of Macroinvertebrate Samplers as They Apply to the Streams of Semi-Arid Regions. In: Proceedings of the Joint Conference on Sensing of Environmental Pollutants, 1978.
11. Armitage, P.D. Downstream Changes in the Composition, Numbers and Biomass of Bottom Fauna in the Trees Below Cow Green Reservoir and in an Unregulated Tributary Maize Beck, in the First Five Years After Impoundment. Hydrobiologia, 58: 145-156, 1978.
12. Lenat, D.R., D.L. Penrose, and K.W. Eagleson, Biological Evaluation of Non-Point Source Pollutants in North Carolina Streams and Rivers. Biological Series No. 102, North Carolina Department of Natural Resources and Community Development, Raleigh, North Carolina, 1979.
13. Wilhm, J.L. Range of Diversity of Index in Benthic Macroinvertebrate Populations. Journal Water Pollution Control Federation 42: R221, 1970.
14. Hilsenhoff, W.L. Use of Arthopods to Evaluate Water Quality of Streams. Technical Bulletin No. 100, Department of Natural Resources, Madison, Wisconsin, 1977.
15. Duda, A.M., D.R. Lenat, and D.L. Penrose, Water Quality Degradation in Urban Streams of the Southeast: Will Non-Point Source Controls Make any Difference? In: Proceedings of the International Symposium on Urban Storm Runoff, Lexington, Kentucky, 1979. pp, 151-159.

DISSOLVED OXYGEN IMPACT FROM URBAN STORM RUNOFF

by

Thomas N. Keefer, Robert K. Simons, and
Raul S. McQuivey
The Sutron Corporation
1925 North Lynn Street
Suite 700
Arlington, Virginia 22209

ABSTRACT

Records from 104 water quality monitoring sites throughout the country were considered to determine whether a correlation exists between strength of dissolved oxygen (DO) deficit and the presence of rainfall and/or storm runoff in and downstream of urban areas.

Sites were screened from more than 1000 monitors maintained by federal and state agencies such as the U.S. Geological Survey (USGS), Environmental Protection Agency (EPA), Ohio River Valley Sanitation Commission, and Wisconsin Department of Natural Resources. Daily data were obtained and processed for 83 of the 104 candidate sites. Of the 83 monitors considered, 42 percent or about one monitor in three of the 104 candidates demonstrated a 60 percent or greater probability of a higher than average DO deficit occurring at times of higher-than-average streamflow or on days with rainfall. This result was obtained by considering daily data for the entire water years. Not all years at any given station exhibited a 60 percent probability; one to three years out of five is typical. The DO levels fell to less than 75 percent saturation at most of the sites where 60 percent of greater probability existed. Levels of 5 mg/l or less were not uncommon.

Detailed hourly data analysis was made at 22 of the sites, with high correlation between flow and DO deficit. Of the 22 monitors examined, 11 would not meet a 5.0-mg/l DO standard; six of the 11 would not meet the EPA-suggested standard of 2.0 mg/l for 4 hr. Streeter-Phelps analysis indicated that two additional monitor sites at which hourly data were examined would not

have met the EPA standard had they been properly located. An additional two sites at which hourly data could not be obtained would also not have met the EPA standard.

In general, the data examined here indicate that 19 percent of the 104 candidate monitors might not meet a 5.0-mg/l standard and 15 percent might not meet a 2.0-mg/l standard. Frequency of violations was not tabulated exactly but appears to be zero to five times per year at sites with correlations.

The data base from which this study was drawn is not geographically homogeneous. No conclusions can thus be drawn on the national scope of either correlations or standards violations. The frequency of EPA standards violations appears to be low, but the causes should be identified to prevent an increase.

INTRODUCTION

The research presented here was designed to determine from existing data bases the behavior of dissolved oxygen (DO) levels downstream of urban areas after storms. The information presented is taken from the project final report, Reference (1).

Continuously recorded water quality information was obtained from data banks such as U.S. Geological Survey (USGS) and the U.S. Environmental Protection Agency's (EPA) STORET system. Data from other state and local agencies such as the Ohio River Natural Resources (WDNR), Greater Metropolitan Chicago Sanitation District, and others were also considered when possible. The data bases were reviewed to obtain a five-year (if possible) historical picture of daily variations in DO, temperature, and flow in the receiving waters of the United States at the location of all reliable continuous water quality monitoring stations. Concurrent with this review, rainfall data for the areas in which the monitoring stations are located were obtained from the National Climatic Center in Asheville, North Carolina. The DO and temperature records were examined to find all locations downstream of urban areas in which DO deficits during the warm-weather portion of the year were correlated with rainfall and/or urban runoff events. At these locations, an analysis was made to determine whether a correlation exists between wet-weather discharge and DO sag on an hourly basis and how significant the wet weather impact is.

Existing monitors are not necessarily located to study urban runoff. The location of the monitoring stations and rain gages relative to the urbanized areas and the relative sizes of urban and nonurban drainage areas contributing to the receiving water were considered in making judgments about urban rainfall/runoff and DO deficits.

OBTAINING DATA

AVAILABLE DATA BASES

A number of federal and state agencies maintain water quality data bases. Those that were used or seriously considered for use in this study are briefly discussed below. Idiosyncrasies of the data bases, if any, are mentioned. After describing the various information banks, the screening procedures used to select monitor sites for this study will be described.

Office of Water Data Coordination (OWDC)

The OWDC is part of the Water Resources Division of the USGS in the U.S. Department of the Interior. It compiles a catalog of water data collection stations throughout the United States. Water quality and quantity monitors for streams, lakes, reservoirs, estuaries, and ground water are included. More than 350 non-federal (state and local) agencies are listed, plus all the relevant federal agencies. The OWDC catalog was a primary source of information for this study. The OWDC does not collect or disperse data. It merely keeps track of those who do. Upon request, it will supply names and addresses of all the agencies listed as well as a personal contact and telephone number if known. The OWDC information, however, is not stored in computer data banks. In order to locate the continuous monitors, it was necessary to go through its 21 volumes one page at a time and decipher the codes.

National Water Data Exchange (NAWDEX)

The NAWDEX is also a program of the Department of the Interior administered through the Water Resources Division of the USGS. While OWDC is a cataloging agency, NAWDEX actually deals in data. When fully operational, NAWDEX will be capable of searching the complete USGS water data records as well as the EPA STORET data base.

At the time the present study was undertaken, the NAWDEX program was only capable of scanning the USGS records. A list of 45 continuous water quality monitors at which flow data were available was obtained. It was subsequently determined that these could have been found in the OWDC catalogs.

USGS Automatic Data Processing (ADP) Unit

The USGS ADP Unit is located at the national headquarters in Reston, Virginia. Its purpose is to provide standardized processing programs and procedures for the water resources data collected by USGS. It is the most up-to-date source of information on the contents of the USGS water quality and flow data base. It is also the place to contact to obtain data in digital format such as cards or magnetic tape. For very reasonable prices ($35 to $70) any of the daily parameter values stored by the ADP Unit can be obtained along with excellent instructions on how to read and translate the cards or tape. The USGS ADP Unit provided a bulk of the data analyzed in this study.

EPA STORET System

STORET is a computerized water quality data base that was conceived and initiated in the early 1960s by the U.S. Public Health Service. Its sole purpose is to allow storage and retrieval of water quality information by government users.

More than 1100 water quality monitor files are included in STORET records. Approximately 200 of these monitors were located on lakes, more than 100 in pipes, almost 100 beside the ocean, and about 50 in intakes and estuaries. This left about 500 monitors located along rivers. Data from all the river sites below urban areas were transferred to a digital tape for use in this study.

The water quality data in STORET is in an unusual format. Daily and hourly observations are mixed together in a continuous string. The potential user should be aware that STORET contains no flow information.

State and Local Agencies

For two reasons, little actual contact was made with state and local agencies during the course of this study. First, the OWDC catalogs are a definitive source of which state and local agencies collect water quality data. The very great percentage of those that do, concentrate on random observation and do not have continuous data bases. Very few data on flow are available from such sources. Second, those state agencies that actually collect continuous data report the data to the STORET data base. Thus, it was not necessary to contact many places. Two local agencies that maintain their own data bases are the Wisconsin Department of Natural Resources and the Metropolitan Sanitation District of Greater Chicago.

Wisconsin Department of Natural Resources--

The WDNR maintains a network of 11 water quality monitors. It also records flow continuously. For a very reasonable amount, it will copy all 11 stations of daily data onto a magnetic tape. Hourly data are also available.

Metropolitan Sanitary District of Greater Chicago--

The Chicago sanitation district maintains a number of flow-water quality monitor stations in and around Chicago. Many are located in canals. Hourly data are available, but not in digital form. A letter from a government project officer explaining the need for the data is required. Because of the manpower and travel requirements, no data were used from this source in this study.

Ohio River Valley Water Sanitation Commission (ORSANCO)--

The ORSANCO is an interstate compact agency. It was created in 1948 by the states bordering the Ohio River primarily to monitor and prevent pollution along the river. To achieve this, ORSANCO maintains a network of water quality monitors, which reports continuously to the headquarters in Cincinnati, Ohio, where a complete picture of the quality of the river at any time is available.

At various times, 56 monitor sites have been placed on the river. Half of these have been in continuous operation since 1961; the DO and temperature data from these stations are available on an hourly basis. However, no flow data are available.

While the data are readily available, there are serious drawbacks, the most important of which is the data format. Because of ORSANCO's interest in an "overall" look at the basin, the numbers are not grouped by station. Instead, each tape record contains data for one day for all stations reporting that day. The records are random length and structured in such a way that hundreds of tape reads are required to sort the data by station. Processing cost was so high for the ORSANCO data that none were included in this report.

National Climatic Center

Precipitation data for the present study were obtained from the National Climatic Center in Asheville, North Carolina. For this study, only two forms of data were used. In the initial phases, a digital tape containing daily precipitation and other variables was obtained. This type of data is very costly. The hourly data for the later phases of the study were obtained from free copies of *Local Climatologic Data,* a monthly publication, obtained from the National Weather Service (NWS) in Silver Spring, Maryland. These publications contain the hourly precipitation for the month of issue.

SITE SELECTION PROCESS

The process of selecting monitors for inclusion in the study consisted of four steps. First, the data bases were scanned to determine which of the water quality stations were actually continuous monitors. Next, the continuous monitor sites were located and each monitor was classified as to its usefulness to this study. Finally, those with the best classification ratings were selected and data obtained.

Monitor Identification

The process of identifying monitor data for use in this study was complicated by the number of agencies involved and the variety of cataloging systems. Many of the possible sources of information (exclusive of the large federal groups such as USGS or EPA) were not known at the outset.

The first government agency contacted was the Water Resources Division of the USGS. The NAWDEX provided a list of continuous USGS water quality monitors where flow was also recorded. Further inquiries led to the Office of Water Data Coordination and individual USGS district offices in the various states.

The OWDC was the best source of monitor sites and produced a very comprehensive list of likely sites. Each of the several thousand pages could contain from zero to 10 monitors useful to the study.

Inquiries concerning USGS monitoring activities were initially directed to district offices in each state. These inquiries led to the Automatic Data Processing Unit in Reston, Virginia. The ADP unit provided a list of all water quality monitors maintained by USGS as well as flow stations. The list was far more comprehensive and up to date than the one obtained from NAWDEX. No further searches were conducted for USGS monitors.

No difficulty was encountered in determining how many monitors were available from ORSANCO or EPA. ORSANCO merely provided a comprehensive list. Cross-checks verified that the same information was available through the OWDC catalogs. At first, it was difficult to gain access to the STORET system. Once a user number was obtained, very little time was required to obtain lists of the monitors contained in the data base. A great many of these were state and local agency monitors that were also listed in the OWDC catalogs.

In all, more than 1500 monitors that recorded continuous DO and temperature plus other parameters were located. The next steps were to locate, classify, and sort them to determine which were near urban areas and had flow and rainfall data available.

Monitor Site Location

Most monitors were located by means of latitude and longitude coordinates. All the station lists with the exception of ORSANCO reported an accurate latitude and longitude for each site; ORSANCO locates its stations by U.S. Army Corps of Engineers (COE) river mile markers. USGS 7.5-minute topographic sheets were used to locate monitors and flow gages. Each monitor was found and its proximity to an urban area noted. A number of monitors were rejected from consideration at this point if no urban area was in a position to affect the stream. Some monitors were located by using the map library at USGS headquarters.

Monitor Site Classification

In classifiying the monitor sites for suitability, the following factors were considered:

- proximity to urban drainage area,
- presence of continuous record,
- length of record,
- proximity of flow gage,
- proximity of rain gage, and
- length of time monitor was in operation.

Several hundred monitors were considered, which made it impossible to remember even a small portion of the information. A standard classification sheet was developed and filled out at the time the monitors were being located. Based on the above factors, a complete list of slightly more than 100 monitors was compiled and considered for analysis.

Summary of Available Data

Slightly more than 40 USGS monitors were found in the NAWDEX system; a search of the OWDC catalog series added 250 more; several additional USGS monitors were located through the ADP Unit; ORSANCO added 56; and a search of the STORET files produced 1100. Thus, nearly 1500 monitor sites were found along the nation's streams, lakes, and estuaries. Each monitor was then located on USGS maps and evaluated for proximity to urban areas. They were further rated on proximity to rainfall and flow gages and for length of record. Those in proximity to an urban area of significant size and were near a flow gage or primary weather station were selected for daily correlation analysis. The number of such sites was slightly greater than 100.

Of 1500 monitors, less than 10 percent were considered. The most common reason for excluding a monitor was that it was not in or near an urban area. The vast majority of EPA monitors (at least those in the STORET files) are at industrial outfalls or in canals or lakes; this is also true of the USGS and ORSANCO networks. A small number of monitors was rejected because no flow records were available. Others were rejected because an examination of the daily monitor records indicated no quality problems. Finally, some sites were excluded because they were incorrectly listed as continuous monitors.

DATA ANALYSIS

DAILY CORRELATION ANALYSIS

The daily correlation analysis was conducted in three distinct phases. First, the daily data from selected sites had to be acquired and converted to a standard format and stored in a computer direct-access file. Second, the stored values had to be retrieved and processed with suitable correlation programs. Finally, sites for hourly and detailed analysis were selected. Each phase will be discussed in turn.

Acquiring and Storing Data

Four major computer programming efforts were required to complete the first phase of data processing. Each type of taped data (USGS, NWS, STORET, and WDNR) required different recovery and processing methods.

USGS Data--

In this study, USGS data were extremely important. When the site selection process was complete, 62 of its monitors were of interest. USGS flow data were often required to supplement DO data from other sources. For these reasons, the USGS data were obtained and processed first. The USGS ADP unit is highly cooperative and accustomed to dealing with the public. Data requests were filled promptly and at reasonable cost. Anyone familiar with programming can have a program to read the data operated in one day. Daily data were transferred to direct access files in USGS format for use in correlation programs.

National Weather Service (NWS) Data--

NWS data are provided on digital magnetic tape as card images. No great difficulty was encountered in reading the tapes. Data were converted to USGS format for convenience in processing.

STORET Data--

STORET data are stored in integer format similar to NWS data. Retrieval was simplified by programs provided by the STORET users center. Daily data were sorted form the mixed files of daily and hourly data and stored in USGS format.

WDNR Data--

The WDNR data were also supplied on digital magnetic tape. The format, which was similar to NWS data, was converted to USGS type as the data were transferred to direct-access files.

Data Retrieval and Correlation

The first step in the daily correlation analysis after the required data were stored was to develop a suitable correlation technique. The objective of the correlation program development was to build a program that was a sensitive indicator of the problem in question. That is, it should give a strong numerical indication whenever a site was found at which the DO level dropped with increasing flow.

The flow and DO signals of the type of data typical of daily analysis are illustrated in Figure 1. They have been plotted to illustrate the correlation of high flow and low DO at Wilson Creek. The problem was to detect this correlation mathematically.

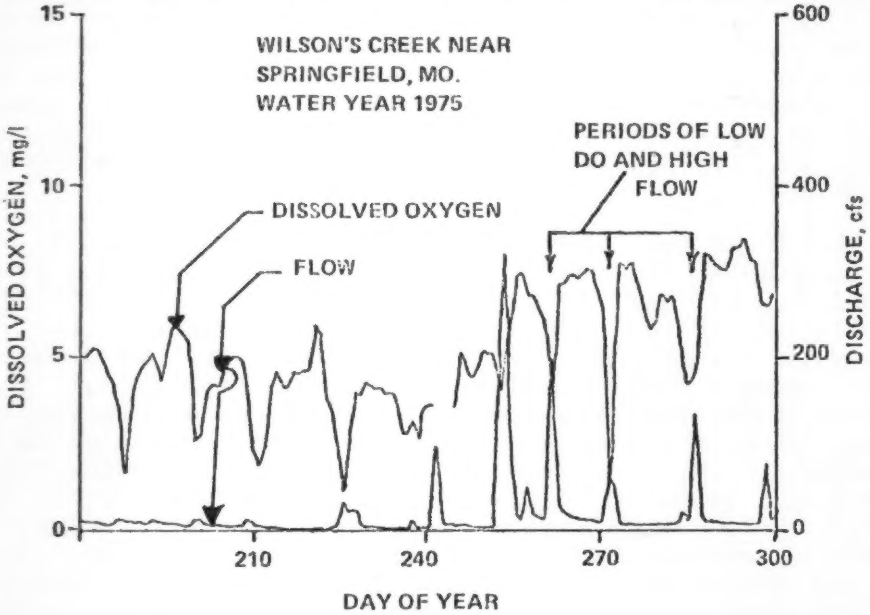


Figure 1. TYPICAL DAILY FLOW AND DO RECORDS ILLUSTRATING HIGH DEGREE OF CORRELATION BETWEEN HIGH FLOW AND LOW DO

Standard statistical cross-correlation proved unworkable for daily data. The DO signal is more or less random in time with some long-term (low-frequency) components. The flow is an intermittent highly skewed signal. Low correlations (0.6 or less) were obtained at all reasonable time lags for the Wilson Creek data. This indicated that other methods had to be developed.

The next correlation method used was based on a comparison of the DO levels on "wet" and "dry" days. Various definitions of "wet" and "dry" were used with both rainfall and flow data. The different definitions of "wet" and "dry" are described as follows. When correlating the DO sag with rainfall, a day was considered dry when the daily precipitation was zero and wet when the rainfall was greater than zero. When correlating the DO sag with flow, a day was considered dry when the discharge was less than a given percentage of mean annual discharge and wet when the discharge was greater than the same percentage of mean annual discharge. The percentage was varied from 50 percent to 400 percent in 50 percent increments in order to determine the effect of the wet/dry cutoff. A wet/dry cutoff equal to the mean annual discharge was found to be a reasonable value.

The method was implemented on a single year of data at a time. First, all the DO deficits were summed for wet days and for dry days. (DO deficit is the difference between the actual DO level and the saturation DO level for a given water temperature.) The number of wet and dry days was also computed. The average dry (wet) DO sag was then calculated by dividing the sum of all dry (wet) DO sags by the number of dry (wet) days. The average annual wet DO sag was then divided by the average annual dry DO sag. If this ratio was greater than unity, the DO sag was worse during wet weather. This method proved sufficiently sensitive to indicate a problem at Wilsons Creek and was retained for use in the analysis. It was not, however, considered the best method.

The final and most sensitive correlation measure was developed by viewing the occurrence of a correlation in a probabilistic sense. Two questions were asked: (1) "Is it more likely that the DO deficit will increase or decrease when the flow changes or a rainfall event occurs?" and (2) "How does this likelihood compare during periods of low and high flow?" The answers to these questions were found by developing a computer counting scheme. Figure 2 illustrates what may be thought of as a four-compartment box. The compartments are labeled A through D and numbers are assigned to the four compartments by graphing DO deficit divided by a moving average deficit versus discharge divided by average discharge. The averaging period was initially not specified, and various values were tried in order to find one most suited to this study. Seven days was ultimately chosen as the best averaging period.

The reasoning behind the graph is as follows. First, some means was necessary to determine whether the DO deficit on any given day was "better" or "worse" than on previous days. The

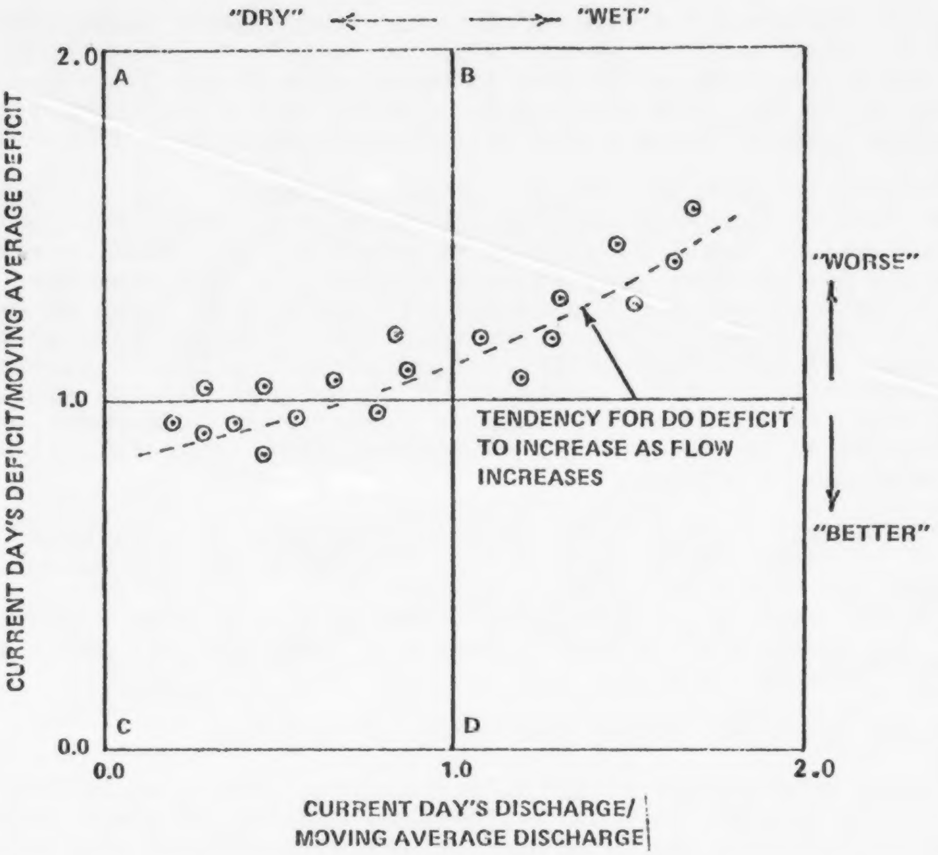


Figure 2. BOX OR COMPARTMENT METHOD USED TO ANALYZE DAILY DATA FOR CORRELATION BETWEEN HIGH FLOW AND LOW DO

method selected compares the value observed "today" with the average of the values that occurred for the previous seven days. Thus, a deficit/moving average deficit value greater than 1.0 indicates "worse" and a value less than 1.0 indicates "better." Points on Figure 2 that lie in the upper half (compartments A and B) represent days when the water quality was worse than it had been for the previous seven days. Points that lie below the line (compartments C and D) represent days on which the water quality improved.

Next, some means was required to determine whether the water quality consistently decreased on days with rainfall or increased flow. Identifying days with rainfall was easy; identifying days with increased flow was again done by moving average. The average daily flow "today" was compared with the average for the previous seven days. A flow/moving average flow value greater than 1.0 indicates "increased" flow or "wet" and a value less than 1.0 indicates "decreased" flow or "dry." Points on Figure 2 that lie to the left of center (compartments A and C) represent days on which the flow decreased. Points that lie to the right of center (compartments B and D) represent days with increases.

The logic of Figure 2 can now be discussed. For any year DO and flow or rainfall data up to 358 points (365 - 7 used for moving average) could be plotted on a graph. By counting the number of points that fell in each of the four compartments, it was possible to compare the number of days during which DO levels decreased and flow increased (or rain fell) (Compartment B) with the number of days when DO levels increased and flow increased (Compartment D). If the contents of Compartment B were significantly greater than the contents of Compartment D, a correlation was considered to be established between flow (or rainfall) and decreased DO.

The probability method was tested on data from Wilsons Creek near Springfield, Missouri. The method was sensitive enough to indicate the known correlation at that site. A greater than 70 percent probability of low DO at times of higher than average (seven-day moving) flow and on days with rainfall was found for several years of data.

Selecting Sites for Hourly and Detailed Analysis

After determining that the probability method would identify a site with a known correlation between flow or rainfall events and DO deficit, it was necessary to set criteria for selecting sites for more-detailed examination. Many sites exhibited correlation between low flow and DO deficit as well as high flow and DO deficit. Only those sites at which low DO could clearly be identified with high flow were desired.

A cutoff level of 60 percent was finally selected. That is, only sites at which the probability of a greater than seven-day moving average DO deficit at times of greater than seven-day moving average flow were chosen for further study. The 60 percent cutoff level can be verified as being statistically valid. A Chi-squared test was used to demonstrate that a probability of greater than 60 percent or less than 40 percent is required in order for nonrandom distributions to exist between two categories at the 95 percent confidence level. Thus, for a given station year, if a 60 percent probability of low DO exists at times of high flow or on days with rainfall, low DO is significantly associated with these events.

Consideration was also given to the absolute DO level in selecting sites. Even though a correlation existed between low DO and flow or rainfall, a "problem" did not necessarily exist.

In general, sites were selected where DO levels less than 75 percent of saturation were present at times of high flow.

Results of Daily Correlation Analysis

Presentation of the results of the daily analysis is complicated by the volume of numbers involved. Some 104 potential locations were to be analyzed. Three primary factors reduced the number to 83. First, data could not be obtained for some sites; second, information in the OWDC catalog was not always accurate; and finally, the list of 104 sites included a number of ORSANCO monitors that were not analyzed because of the difficulty with the data. The final list of 83 sites includes 55 USGS monitors, 17 STORET monitors, and 11 WDNR monitors. Periods of record range from one to five years.

Daily rainfall correlations were completed at most of the USGS monitor sites. The prohibitive expense of the National Climatic Center data discouraged the completion of the rainfall correlation at the STORET and WDNR sites.

From the 83 monitors examined on a daily basis, the following results were obtained:

- in 19 USGS stations, low DO correlated with flow;
- in 2 EPA-STORET stations, low DO correlated with flow;
- in 3 WDNR stations, low DO correlated with flow; and
- in 18 USGS stations, low DO correlated with rainfall.

The stations in the above group were examined closely, and hourly data were obtained for further analysis when possible.

DETAILED SITE ANALYSIS

General Considerations

Water quality monitor sites at which correlations existed between rainfall and/or flow and low DO were identified. The daily data used to identify sites did not allow a determination of specific water quality standard violations. Hourly data analysis was required at sites that showed strong deficits (DO levels less than 75 percent saturation). For these reasons, it was necessary to examine those sites with correlations in greater detail.

Streeter-Phelps analysis was performed at several selected sites to determine a given water quality monitor was in a position to sense the maximum deficit from urban runoff and whether observed deficits appeared reasonable based on the best estimates of input loads. In addition to selecting sites for hourly and Streeter-Phelps analysis, it was desired to form some opinion on why a correlation might exist between flow/rainfall and low DO at a given site. Information was needed on travel time from the urban area to the monitor, distance from the rain gage to the urban area, and other factors that might influence the DO tha a monitor "sees."

Analysis Procedure

The first step in the detailed site analysis was the gathering of available maps. The monitor and flow gage and the weather station, if any, were then located precisely. An information sheet that summarized what could be learned from USGS records, the map, or other sources was next filled out. Information sheet data were used to select sites with the simplest hydraulic conditions for anlaysis by the Streeter-Phelps technique. Finally, the plots of daily average flow, DO, and rainfall values were examined, and periods for hourly analysis were selected. From 10 to 30 days or more of hourly data were examined at each site at which the hourly data could be obtained. The total of all this information was then studied carefully to see what could be concluded as presented in the results reaction.

Obtaining and Analyzing Detailed Information--

The gathering of information at each site was systematized by developing a standard form. The first step in filling out the analysis forms was usually to assemble USGS 7.5-minute topographic sheets to form a compelte picture of the particular urban area in question. Next, the water quality monitor, the stream gage, and the precipitation station were located by plotting the reported latitude and longitude of the facilities. This had been done in the early phases of the study to select monitors for inclusion in the daily analysis. Many of the maps used, however, were in the USGS library; more maps were purchased and brought to Sutron for the detailed analysis work.

The assembled maps were large and cumbersome. Therefore, a sketch of relevant features was transferred to the site form. This procedure aided in remembering each one. Any outstanding features, such as industry or sewage treatment plants, were noted. Other information gathered from the maps included the urban drainage area, stream width, the distance from monitor to stream gage, and distances from the monitor to sewage outfalls. The urban area was estimated as accurately as possible from the rose-tinted areas on the maps. Distances were measured by map scale and dividers.

After tabulating the physical features of the site, the stream flow was examined. The flow and DO records were gathered and the range of values entered. The plots of DO and discharge were examined to determine the percentage of missing records. This examination was combined with the quality estimates in the agency reports to rate the record. Records were "excellent" if only one or two weeks a year were missing; a "good" rating went to stations with one or two months missing; and "poor" rating went to all others.

While examining the flow records, typical July-August low flow values were selected for later use in the Streeter-Phelps analysis (if performed). If rating curves were available, they were used to determine the depth and velocity, which were also required for Streeter-Phelps.

The results of the daily correlation analysis were summarized to give quick recognition of the magnitude of the "problem" at each site. This was omitted in cases where there was too much information to fit and a note was entered referring to a separate summary table.

The population figures were obtained from a World Almanac from the 1970 census and no attempt was made to account for growth increases. The maps were often more out of data than the census. For consistancy, no adjustments were made to "urban" areas either.

The Streeter-Phelps portion of the site forms was completed only if the analysis was actually performed. Development of complex water quality models was not within the scope of work. For this reason, sites with fairly simple hydraulic conditions were selected for analysis. Meaningful analysis using the Streeter-Phelps method required streams with few tributaries and not too many waste sources. Although only five sites were required by the contract, 13 sites were used because their use contributed a worthwhile insight into the location of monitors. The bulk of the "Streeter-Phelps" theory used in this study was derived from Thomann's (2) book on water quality management. The complete procedure is described in Reference (1).

Obtaining and Analyzing Hourly Data--

The USGS, ORSANCO, and WDNR all record hourly data, but only USGS hourly data were included in this study. Hourly information is obtained as a biproduct of USGS's archiving of mean daily flow. Primary computation sheets or "primaries" are computer printouts of stage and flow (also DO and temperature) from raw digital data tapes. The primaries are the only record of hourly or bihourly data available. The digital tapes from which the primaries are made are reused, thus destroying the only digital form of the data. Each primary sheet is carefully examined by a hydrologist and periods of erroneous, missing, or shifted data are identified. A second series of programs is then run to create a file of corrected data. The corrected files are then averaged daily and the results stored in the USGS backfile library of daily values.

For this study, the primary sheets were obtained for 22 of the 30 USGS sites that showed positive correlation with rain or flow. For various reasons hourly data could not be obtained at the remaining eight sites. In most cases, the problem was inability to obtain the primary sheets. A number of older stations still record data on strip charts, which are not converted to digital form. In a very small number of instances, the District Chief of a particular state would not release the primaries becuase of alleged deficiencies in the data. Emphasis was placed on obtaining periods of hourly data during June, July, August, and September. In addition to the primary sheets, appropriate rating curves were obtained for use in converting the stage data to discharge. Primary sheets report hourly stage instead of hourly flow.

Recovering useful data from the primary sheets was simplified for this study by developing a flexible plotting and analysis sequence for a desktop calculator. First, a program was developed that created digital magnetic-tape cartridge files. Each day of data of a particular kind (stage, DO, rainfall, or temperature) was entered as a separate tape file. Files were created only for periods when all four types of data were available simultaneously. Next, a processor and plotting program was developed. This program reads the files of data, applies the rating curve to the stage data to obtain flow, calculates the saturation DO level, and plots the results, including rainfall.

Hourly data plots from many of the stations with storm flow/low DO correlations were similar. A one-month period of hourly data for the Scioto River at Chillicothe, Ohio, is illustrated in Figure 3. A portion of the plot of 1972 daily data that contains the month is illustrated in Figure 4. These two plots are fairly typical of the hourly analysis procedure. The correspondence between decreasing DO and increased flow is evident in Figure 4. The upper line in the figure is the observed daily minimum DO level, the lower line is the mean daily discharge. A decrease in the DO level can be seen in almost all instances when the flow increases. The period from Day 215 to Day 245 was selected as typical. Figure 3 provides a highly detailed hourly look at this same time period. The upper graph in Figure 3 illustrates the behavior of the absolute DO level. Daily fluctuations of approximately 4.0 mg/l were common before the rainfall and subsequent runoff event. The upper line in the DO graph is the saturation level based on the hourly water temperature. Periods of supersaturation occurred on Days 3 and 11, and the DO level fell below 2.0 mg/l and remained there for three days beginning at Day 18. This precisely coincides with the peak of

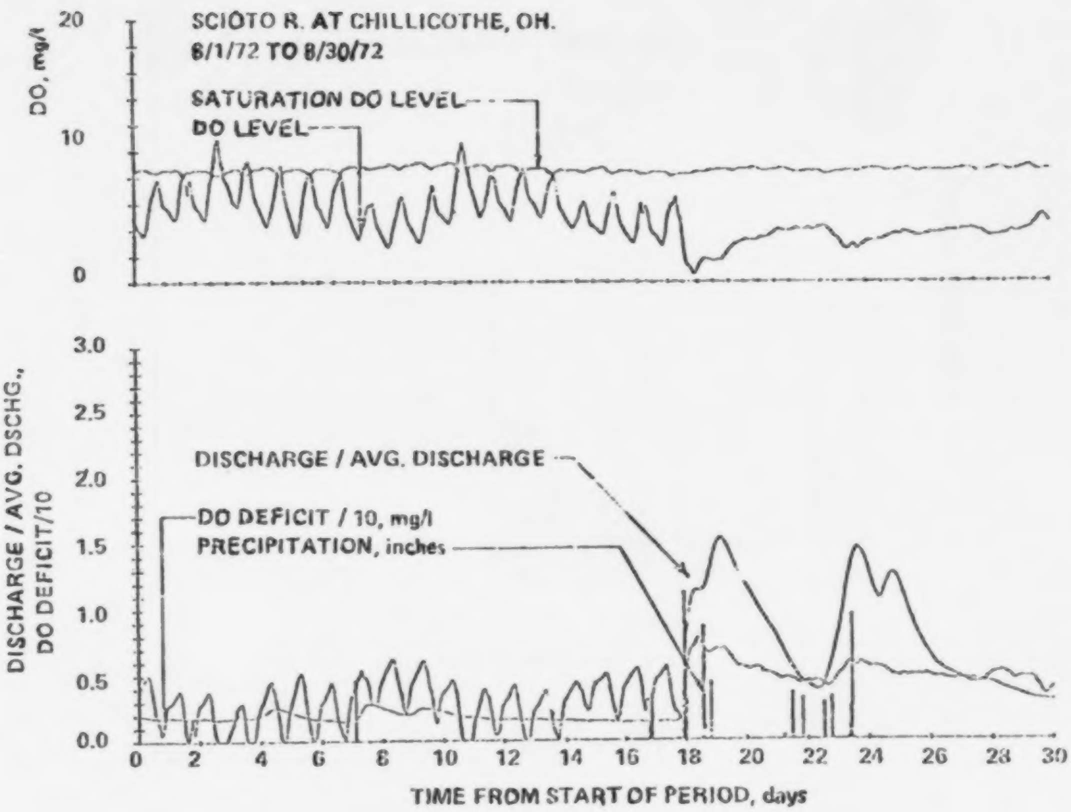


Figure 3. ONE MONTH OF HOURLY DATA FOR THE SCIOTO RIVER AT CHILLICOTHE, OHIO

the flow hydrograph in the lower graph. The lower graph in Figure 3 illustrates the behavior of the flow, rainfall, and DO deficit. The flow has been plotted as discharge divided by average discharge for the period. This was simply a convenience to avoid rescaling every graph. The shape of the hydrograph is perserved, and no useful information is lost. The DO deficit level is the differency between the upper and lower lines in the upper graph of Figure 3. This was included in the lower graph to aid in seeing the coincidence between a change in the behavior of the DO level and the change in flow. Rainfall amounts are plotted as histograms. The period of greater than average DO deficit that occurs during the time of higher-than-average flow can clearly be seen extending from Day 18 to Day 30. There is a clear correspondence between the rainfall events and the increases in flow.

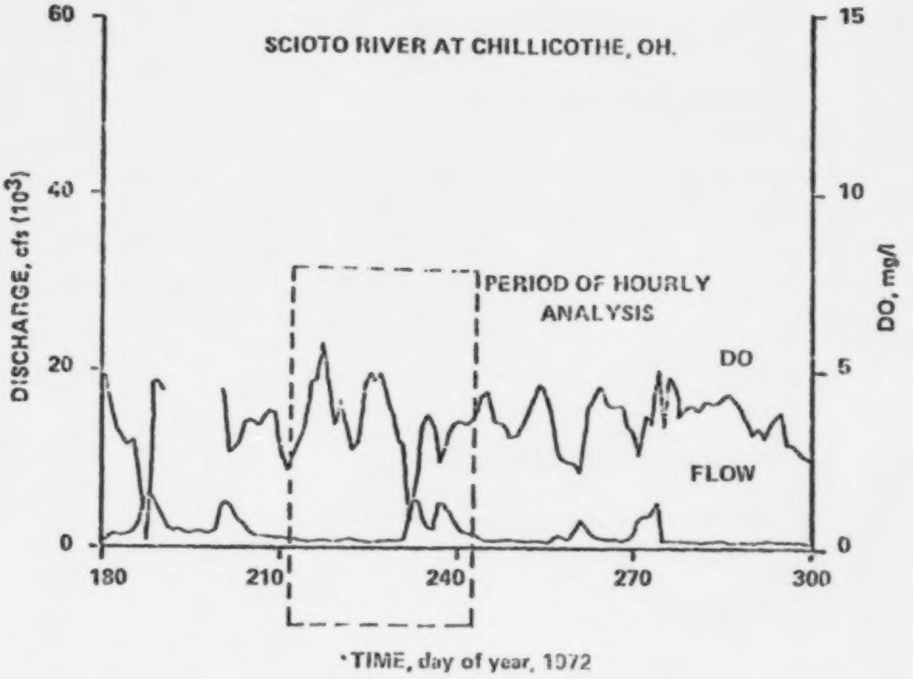


Figure 4. HOURLY DATA FOR ONE-MONTH PERIOD, SCIOTO RIVER AT CHILLICOTHE, OHIO

RESULTS AND CONCLUSIONS

SUMMARY OF RESULTS

Correlation of Low DO with Flow and Rainfall

Daily correlation analysis using USGS, STORET or WDNR flow or rainfall or both was attempted at 104 stations. Of these, 83 - 53 USGS monitors, 17 STORET monitors, and 11 WDNR monitors - had sufficient data to produce results.

Of the USGS monitors, 19 exhibited a 60 percent or greater probability of low DO at times of high flow; 18 exhibited a 60 percent or greater probability of low DO on days with rainfall; and 8 correlated with both flow and rainfall. A total of 30 stations exhibited either type correlation.

Of the 17 STORET monitors, three exhibited a 60 percent or greater probability of low DO at times of high flow. Of the 11 WDNR monitors, three exhibited the 60 percent or greater probability.

Out of 100 candidates for analysis, 36 gave positive results in the correlation analysis, and 17 could not be correlated because of data problems. Thus, 42 pervent of the monitors successfully examined or 36 percent of the likely candidates gave positive correlation results. For discussion purposes, it is thus concluded that one monitor in three placed near an urban area might indicate lower than average DO at times of storm runoff.

Magnitude of DO Deficits

Because of the coarse time resolution of the daily data, no direct conclusions could be drawn about the severity of the DO deficits that sometimes accompany high flow or rainfall. Examination of hourly data at sites with strong daily correlation indicated that water quality violations can occur. Of the 30 USGS sites, 11 clearly indicate a severe DO deficit at times of high flow. Eleven additional sites clearly indicate an effect from storm runoff but could not be classified as severe. Several of these sites were too close to the urban area to detect the maximum deficit in the Streeter-Phelps sense.

Six of the sites did not have water quality problems, at least concerning DO. Some depression during storm events would usually be seen.

Characteristics of DO Variations

Although the 11 sites with severe problems varied widely in size and physical setting, all demonstrated remarkably similar hourly data records. The period prior to a storm event is characterized by fairly large diurnal cycles in the DO level, with 4 to 5 mg/l not being unusual. Periods of supersaturation are often indicated. As the flow increases, the diurnal cycles in the DO record disappear possibly because increased turbidity and depth cuts off the sunlight to the aquatic plant life at times of high flow.

At the time of peak flow, deficit levels at least equal to and occasionally 40 to 50 percent higher than the peak diurnal cycle value are reached.

The effect of the storm flow on the DO level lasts quite a long time. This long-term effect gives added validity to the daily analysis procedure. Sites at which a single hydrograph peak was examined usually recovered in 3 to 5 days.

Comparison of Storm Flow Deficits and Water Quality Standards

When measured against existing stream quality standards, the DO deficits accompanying storm flow may cause water quality violations. Hourly data at monitor sites with strong correlation consistently show deficits below 5.0 mg/l extending over several days. The Streeter-Phelps analysis of ten sites, including several with strong correlations, consistently indicated that monitors are not ideally placed to sense maximum effects. In many cases, deficits 30 to 50 percent stronger could theoretically be found.

Storm-flow-related deficits at some sites consistently violate 2.0 mg/l 4-hour standards. In fairness, it must be pointed out that at these sites the water quality is marginal at all times. Storm events merely push the level down further. General improvements in water quality at all the critical sites would help alleviate the problem.

National Distribution of Storm Runoff Related DO Problems

The question of whether DO deficits caused by urban runoff is a national problem is difficult to answer. One item to consider is the geographic coverage of this study. The data base maintained by the USGS at its Reston headquarters contains records for 150 water quality monitors. These monitors are located in 30 of the 48 conterminous states for 63 percent coverage. The distribution of states containing monitors is highly nonuniform. Thirteen states are east of the Mississippi River and 17 are west. Only 47 of the 150 monitors are in western states. If the dividing line between East and West is considered to be along the western boundaries of Louisiana, Arkansas, Iowa, and Minnesota, the distribution becomes even more unbalanced. Only 19 monitors are then located in the West.

The distribution of monitors by state is highly nonuniform also. Ohio alone has 32 monitors, followed by New Jersey with 13 and Louisiana with 11.

The existing monitor network is probably inadequate to define a national problem. The probability is approximately one in three of detecting a correlation between flow and DO deficit. Many states with significant urban areas on streams have no monitor records at all in the major data banks.

SUMMARY OF CONCLUSIONS

The following conclusions were reached from this study:

- The probability is approximately one in three that analysis of data from a currently existing water quality monitor in or near an urban area will show a correlation between high flow and/or rainfall events and high DO deficit.

- In close examination of sites that exhibited daily correlation between flow and low DO or both rainfall and flow and low DO, strong visual evidence was usually found of the correlation in hourly data records. Sites that exhibited daily correlation with rainfall seldom showed any visible evidence of correlation in hourly data records. Flow was judged the better correlation parameter because it is a direct rather than a secondary indicator of stream conditions.

- At stations where correlation exists, maximum deficits observed during periods of high flow are equal to or 10 to 15 percent greater than the maximum deficits observed during diurnal cycles at times of steady low flow.

- At locations where water quality is already marginal (e.g., 5-7 mg/l DO), a storm event can result in DO levels less than 4-5 mg/l for periods of several days or longer. Occasional violations of 20 mg/l 4-hour standards occur.

- Most water quality monitors are located too close to the urban areas to detect the maximum possible DO deficit in a Streeter-Phelps sense.

- Flow and DO deficit are correlated in a wide vairety of urban situations ranging from towns of 20,000 population to urban megalopolises of greater than 1 million population.

- The absence of a correlation between flow and DO deficit does not correlate well with the percentage of contributing urban areas in the drainage basin. There are exceptions in extreme cases where 50 percent or more of the contributing area is urban.

- There is no reliable way to extend the results of this study to an estimate of total effective stream miles nationally. No government agency currently published data on the distribution of population along major rivers.

- The probability of a high DO deficit occurring at times of high flow is not demonstratably greater during the summer months.

- In most instances, the exact cause of the increase in DO deficit is not obvious. In some cases, there are heavy concentrations of industry along the stream channel. In other cases, a sewage treatment plant is close upstream. Reintrainment of benthic material is a likely cause in such locations. In other instances, there is no industry or sewage treatment shown on the site maps. Here, the problem may be strictly due to biochemical oxygen demand (BOD) of urban runoff, a chemical oxygen demand, or other problems.

- Violations of water quality standards appear to be infrequent. The data examined here do not seem to indicate a need for panic programs to treat urban runoff. Rather, further research is in order to define and alleviate the causes of occasional problems.

SECTION 5

REFERENCES

1. Keefer, Thomas N.; Simons, Robert K.; and McQuivey, R.S., "Dissolved Oxygen Impact From Urban Storm Runoff," Final Report, EPA Contract No. 68-03-2630, March 1979.

2. Thomann, Robert V., "Systems Analysis and Water Quality Management," Environmental Research Applications, Inc., New York, New York, 1972, p. 13.

THE IMPACT OF COMBINED SEWER OVERFLOWS ON THE DISSOLVED OXYGEN CONCENTRATION OF A SMALL STREAM

by
Thorkild Hvitved-Jacobsen
Department of Environmental Engineering
University of Aalborg, Denmark

ABSTRACT

Investigations were run in order to determine the impact of combined sewer overflows on the dissolved oxygen concentration of a small stream. The different oxygen consumption processes in the receiving stream were studied at 4 consecutive stations during and after the passage of the discharged polluted volume.

Two different effects on the DO-concentrations were observed in the stream:

1. An immediate effect caused by degradation of mainly the soluble BOD-fraction in the water body and by direct absorption of organic matter by the animals (invertebrates) in the bottom.
2. A delayed effect caused by degradation of the adsorbed colloidal and outsettling particulate matter. After passage of the discharged polluted volume there is still observed an effect on the DO-concentration in the stream. The delayed degradation may increase the respiration of the bottom about 100%. This delayed effect may last 12-24 hours after the discharge event.

The investigation shows that the oxygen consumption of the organic matter input to the stream by a storm overflow due to the delayed effect is spread over a long period compared to time of passage of the polluted volume. This effect appears to be essential for evaluation and design of sewage systems within urban areas especially by discharge of run-off water with high content of BOD.

INTRODUCTION

During the recent years municipal raw wastewater in Denmark has been increasingly treated with special respect to dry-weather situations. Since 1974 these efforts have been supported by the environmental protection act, which states the environmental considerations to maintain a comprehensive flora and fauna.

Under these circumstances the relative effect of storm overflows from urban areas becoming a source of pollution of the aquatic recipients will increase. In Denmark discharge of stormwater from combined sewer-systems to streams is a special, but very important case. The problem is often that a town of 1,000-10,000 inhabitants must discharge municipal wastewater and stormwater to a small stream with a flow of 50-500 ℓ/sec. During intensive storms this dry-weather flow may, for short periods, be increased several hundred percent.

Upon this background an investigation dealing with the impact of run-off water from combined sewers on receiving streams is being sponsored by the Danish Association of Civil, Mechanical, Electrical and Chemical Engineers and financially supported by the National Agency of Environmental Protection, Copenhagen. The hydraulic, physical-chemical and biochemical parts of this investigation are carried out by the University of Aalborg while the biological part is executed by the Freshwater Laboratory of the Agency of Environmental Protection. These investigations are taking place from 1978 to 1981. A provisional report was prepared /1/.

The most important part of these investigations deals with the impact of the receiving stream. The reason why the DO-concentration is of special interest is that this parameter is of great importance for the water quality of a stream. Furthermore the discharged water from combined sewers may be a potential source for undesirable impact on the DO-concentration in the stream.

During the last years in Denmark, as well as abroad, several investigations dealing with the generation of the stormwater together with nature and contents of pollutants in the run-off water have been carried out. Compare as an example survey articles by Harremoes (1977) /2/ and Johansen (1974) /3/.

However, it can be ascertained that only very few investigations deal with the impact of the storm events on the receiving water bodies, Horner et al. (1978(/6/.

The interest of the National Agency of Environmental Protection in having a better understanding of the impact on the receiving streams during storm events is caused by the fact that the actual practice for discharge of run-off water seems to ignore now-a-days demand to water quality in the recipients. This practice mainly considers a hydraulic protection of the sewage treatment plant for continuous optimal function and not a conscious protection of the recipient. The aim is to have this practice replaced by instructive lines for protection of recipients in connection with discharge of stormwater. The performed investigation has to be seen as a step tending towards obtaining the necessary knowledge of the impact on the recipients in order to reach this aim.

INVESTIGATED AREA

The investigation was carried out in Skravad Brook located in the middle of Jutland, Denmark (fig. 1).

The stream receives, about 5.5 km (3.4 miles) upstream the confluence of

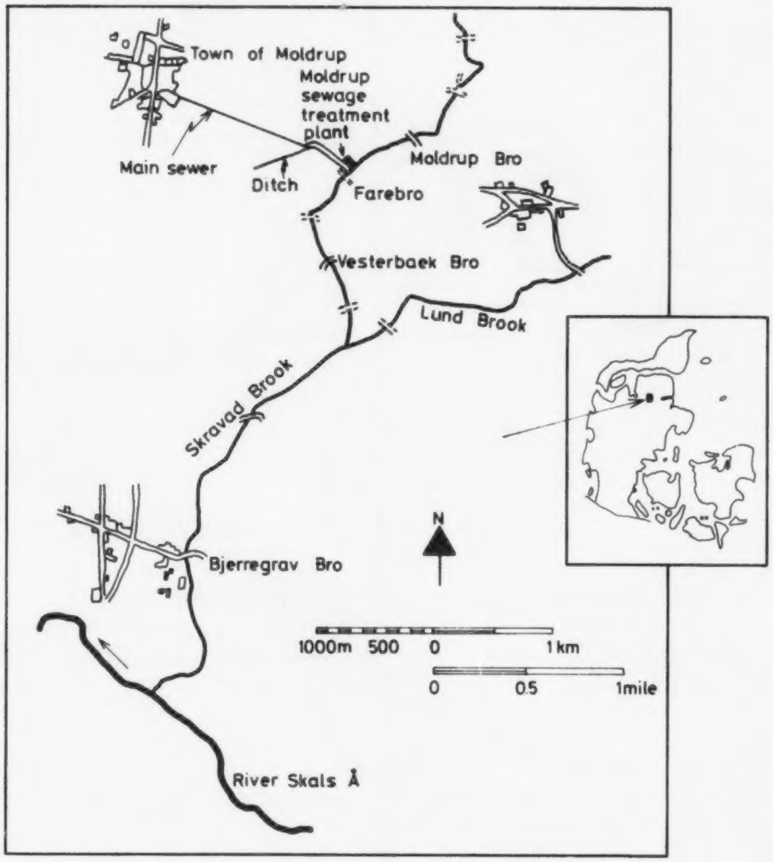


Fig. 1: Map of the investigated area, Skravad Brook.

Skravad Brook and River Skals Å, biologically treated wastewater from Moldrup sewage treatment plant. From the small town of Moldrup which has a combined sewerage system and covers a reduced area of about 15 ha (37 acres) the overflow water is discharged through a ditch to Skravad Brook just downstream the outlet from the sewage treatment plant (fig. 2).

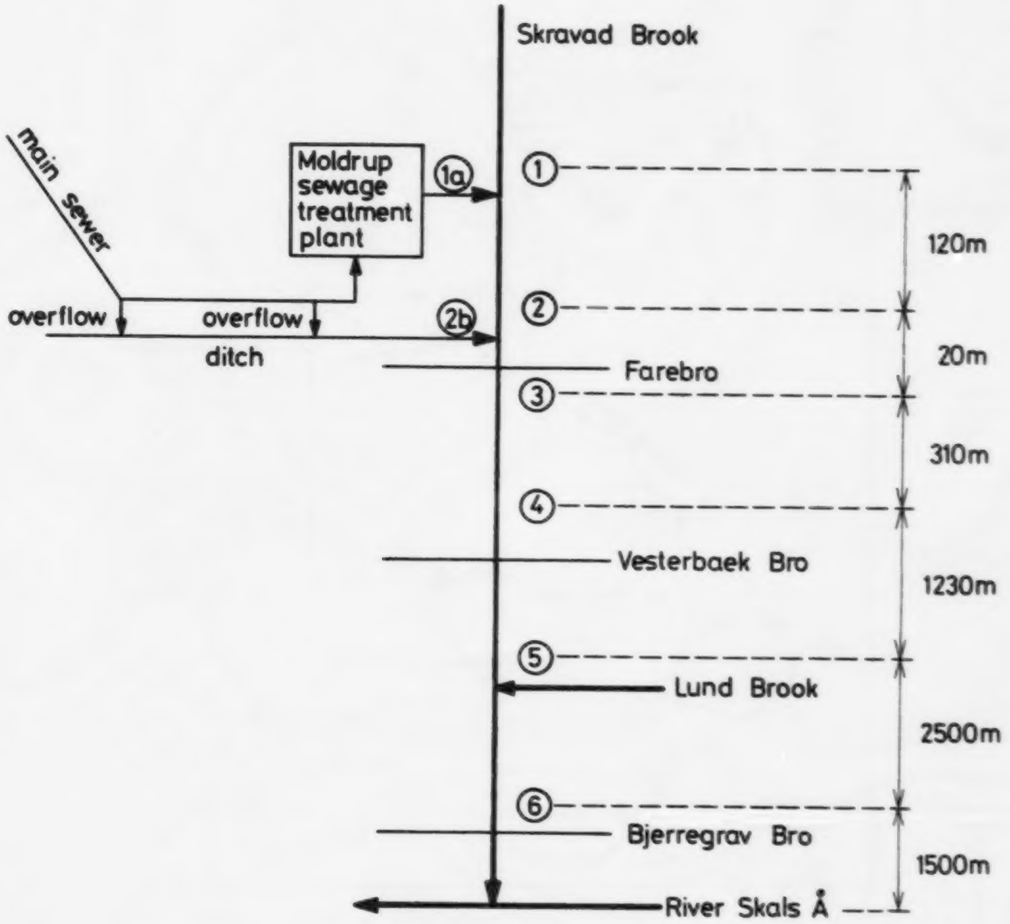


Fig. 2: Schematic representation of the investigation area in Skravad Brook. Stations for sampling and DO-concentration measurements in inlet and recipient are stated.

Moldrup town has about 1,000 inhabitants and includes a few industries, among them a dairy and a small slaughter-house.

Upstream the outlet from the sewage treatment plant of Moldrup, Skravad Brook is influenced by drain and surface run-off from mainly rural areas. The relatively moderate influence on the stream upstream as well as downstream the wastewater treatment plant was an important argument for the choice of Skravad Brook as investigation area.

The investigation was started in autumn 1978. At this time the following conditions were characteristic for the stream (station 5, 2.10.78):

Rate of stream reaeration	$K_2(20) = 7\ \text{day}^{-1}$
Total respiration	$R(tot) = 2.5\ gO_2 \cdot m^{-2} \cdot \text{day}^{-1}$
Production	$P = 0.3$ -"-

These conditions cause little variation in DO-concentration throughout day and night and a relatively high level of DO-concentration. In that case the DO-concentration is 9-10 mg/ℓ throughout day and night with a DO-saturation concentration of 11.0-12.5 mg/ℓ. Even though the stream is influenced by treated wastewater (BOD_5 in the stream is 2-4 mg/ℓ during dry-weather conditions) it is possible to maintain favorable oxygen conditions mainly because of a slope of bottom of about 2% and a daily meanflow at station 5 of about 160 ℓ/sec. (autumn 1978). Apart from periods with extreme growth of aquatic plants in the summer period it is possible to maintain a DO-concentration throughout day and night exceeding 6 mg/ℓ.

With respect to the oxygen conditions the investigated stream can thus be characterized as favorable for maintenance of a comprehensive flora and fauna. Salmon and trout are thus often observed in the stream. The impact of the run-off water on the DO-concentration in the stream is thus expected to be reflected on the background of the recently discussed oxygen conditions during dry-weather situations.

OXYGEN CONDITIONS DURING DISCHARGE OF RUN-OFF WATER

From a theoretical point of view the classical Streeter-Phelps model /7/ is, during steady-state conditions, able to describe the deoxygenation and physical reaeration in streams. Furthermore it is possible to develop the model by considering more and better description of these processes. During dry-weather periods it is often possible to maintain approximately steady-state conditions, and removal of organic matter from the water body will be equivalent to the oxygen demand whether the degradation takes place in the water body or at the bottom. Concerning the deoxygenation at the bottom, it is naturally not the same organic matter, which is actually removed from the water as at the same time interval is degradated at the bottom. As the Streeter-Phelps model is a simple oxygen balance these conditions are left out of consideration and are for the steady-state situations without interest.

During non-steady-state situations, for instance run-off events, concentration of organic matter will be increased in the stream. Removal of organic matter from the water body will be increased too and exceed the actual degradation at the bottom. This fact may result in a later increased degradation at the bottom and a delayed effect on the DO-concentration which may affect the stream after passage of the discharged stormwater volume. Non-steady-state condition thus requires distinguishing between an immediate and a delayed effect on the DO-concentration.

For river systems, influenced by effluents characterized by a BOD-content, the DO-concentration in the stream is of special interest. In connection with combined sewer overflow, change in the DO-concentration in the stream is influenced by:

I. The resulting DO-concentration obtained immediately after mixing of combined sewer overflow and water from the stream.

II. The combined sewer overflow impact on the DO-influencing processes in the stream.

re I. The following phenomena will influence the DO-concentration after mixing storm sewage and river water:

a) DO-concentration in mixed stormwater and raw wastewater before discharge as a result of DO-concentrations in these two components.

b) Flow of combined sewer overflow in relation to flow in the stream. This relation is influenced by time of day and night and time of year.

c) DO-concentration in the river upstream point of discharge.

d) The design of the sewerage system.

re II. The following processes are expected to be mainly responsible for changes in DO-concentration of the stream:

a) Oxygen demand by degradation of organic matter.

A distinction must take place between:

1) An immediate oxygen demand which is supposed to be due to:

-Degradation in the water body.

-Absorption by animals (invertebrates) in the bottom sediment.

Both processes are due to degradation of mainly dissolved organic matter.

2) A delayed oxygen demand caused by:

-Adsorption of mainly colloidal organic matter.

-Degradation of settled particulate matter.

Harremoes 1979 /8/.

b) Oxygen demand by nitrification.

c) Photosynthesis and respiration by the aquatic plants.

d) Reaeration from the atmosphere.

METHODS

The purpose of the investigation was primarily to examine the stream response to the combined sewer overflow. The processes mentioned in the previous section point IIa were therefore of special interest.

It was thus possible to carry out the investigations by means of simulated overflows. The day before the simulated overflow was carried out, riverwater from Skravad Brook and raw wastewater from Moldrup in proportions 10:1 was pumped into a reservoir of 370 m^3 at the wastewater treatment plant of Moldrup. On the day of the simulation this synthetic overflow water was let out into the ditch and from there carried on the Skravad Brook (fig. 2).

The simulations were hydraulically as well as to pollutants carried out at the locality like real overflow situations. The following data characterize the simulated discharge on 9.11.1978:

The duration of discharge:	70 min.
Discharged quantity of water:	215 m^3
Dry-weather flow in ditch:	10 ℓ/sec.
Maximum flow in the ditch during discharge:	110 ℓ/sec.
Flow in Skravad Brook just upstream the outlet from the ditch:	120 ℓ/sec.
DO-concentration in the ditch during the discharge:	4.5-6.5 mg/ℓ
DO-concentration in Skravad Brook just upstream the outlet from the ditch:	10.5 mg/ℓ
Quantity of BOD in the discharged water:	15 kg BOD_5

It was essential to measure the expected immediate and delayed effects on the DO-concentration in the stream caused by the overflow.

Throughout the whole period of investigation it was found that the DO-concentration curves through day and night in dry-weather situations at the stations 2-6 had nearly uniform shapes. This proved very profitable for the evaluation of DO-variations caused by overflow. All the DO-concentration curves will thus be identical, partly by displacement of DO-concentration level, partly by change of time corresponding to the time of travel between the stations concerned.

In order to measure the effect of discharge on the DO-concentration in the stream, the DO-concentration curve from station 2, which can be regarded as unaffected by the discharge, therefore, by comparison with the DO-concentration curves from stations 3-6, states the impact at these stations during and after passage of discharged watervolume.

As an example fig. 3 shows the DO-concentration before, during and after the discharge on 28.9.78. At this figure the solid line shows the measured DO-concentrations while the broken line (the displaced curve from station 2)

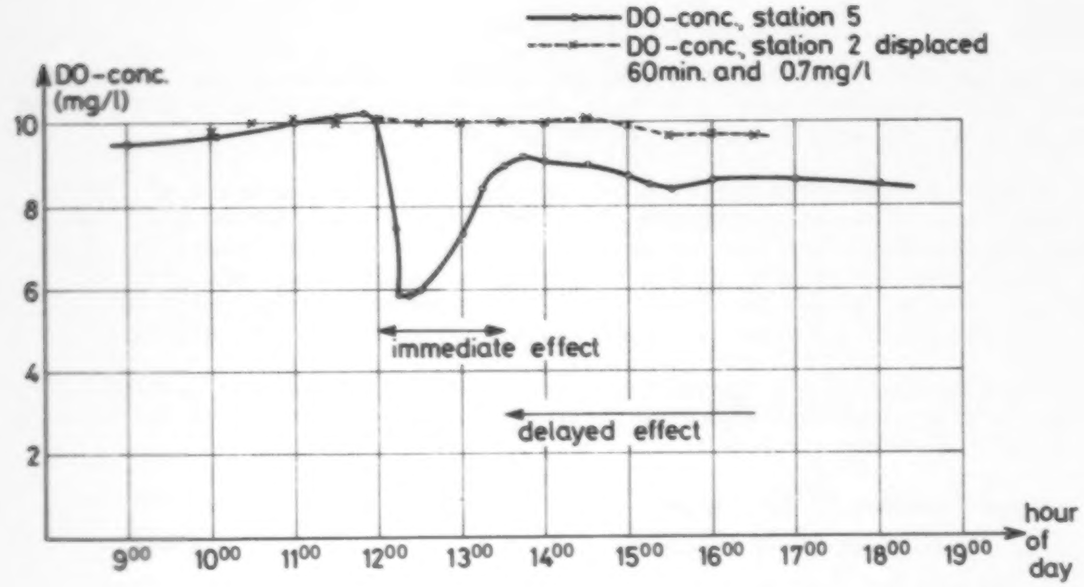


Fig. 3: DO-concentration at station 5 before, during and after discharge on 28.9.78 compared to expected DO-concentration if no discharge took place (displaced curve from station 2)

indicates which DO-concentration may be expected in station 5 if no discharge took place. The difference between such two curves for each station will thus at any time state the effect of the discharge on DO-concentration at the station concerned.

IMMEDIATE EFFECT IN THE STREAM

As earlier mentioned the immediate effect on the DO-concentration in a stream will take place when the discharged polluted watervolume moves downstream. Degradation in the water body as well as absorption at the bottom may influence the DO-concentration.

Fig. 4 expresses the stream response on a simulated discharge carried out in the period from 12:00 to 15:30 on 8.11.78. The discharge was hydraulically moderate as maximum flow in the ditch was 50 ℓ/sec. and only for half an hour the flow was greater than 20 ℓ/sec. The DO-concentrations at station 3 shows the effect on the stream as the discharged overflow by outlet from the ditch is mixed with the stream water. The flow in the stream before discharge was 120 ℓ/sec.

The immediate oxygen consumption which occurs at the 2.5 miles distance from station 3 to 6 can e.g. be recognized by following the observed two minima on each of the DO-concentration curves. These minima appear at 13:00 and 14:50 at station 3 and are observed at 15:10 and 17:00 at station 6. Com-

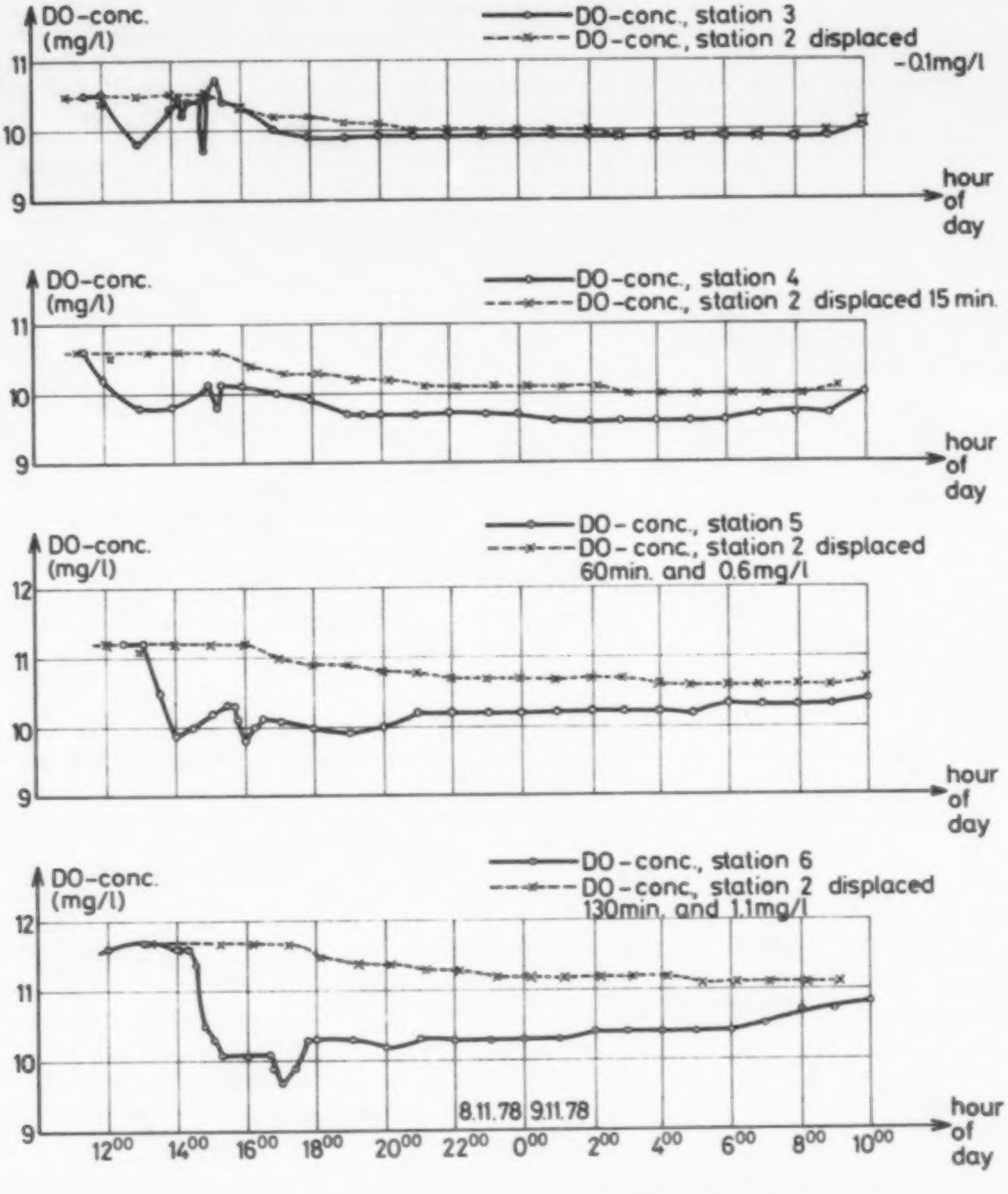


Fig. 4: DO-concentrations at the stations 3 to 6 before, during and after a discharge compared to expected DO-concentration if no discharge took place (displ. curve from st. 2).

pared to the expected DO-concentration expressed by displacement of the DO-curve from station 2, the effect of the immediate oxygen consumption is observed. In the investigated stream with good atmospheric reaeration it cannot be expected that a low value of DO-concentration caused by discharge of combined sewer overflows can be maintained or developed. The following day on 9.11.78 where a greater decrease in DO-concentration was generated by a simulated discharge, it was observed that there could not be obtained greater difference between actual and normal expected DO-concentration than about 2 mg/ℓ at station 6 (fig. 5). These facts were connected with a BOD_5 mean value in the stream of about 20 mg/ℓ during passage of the polluted watervolume.

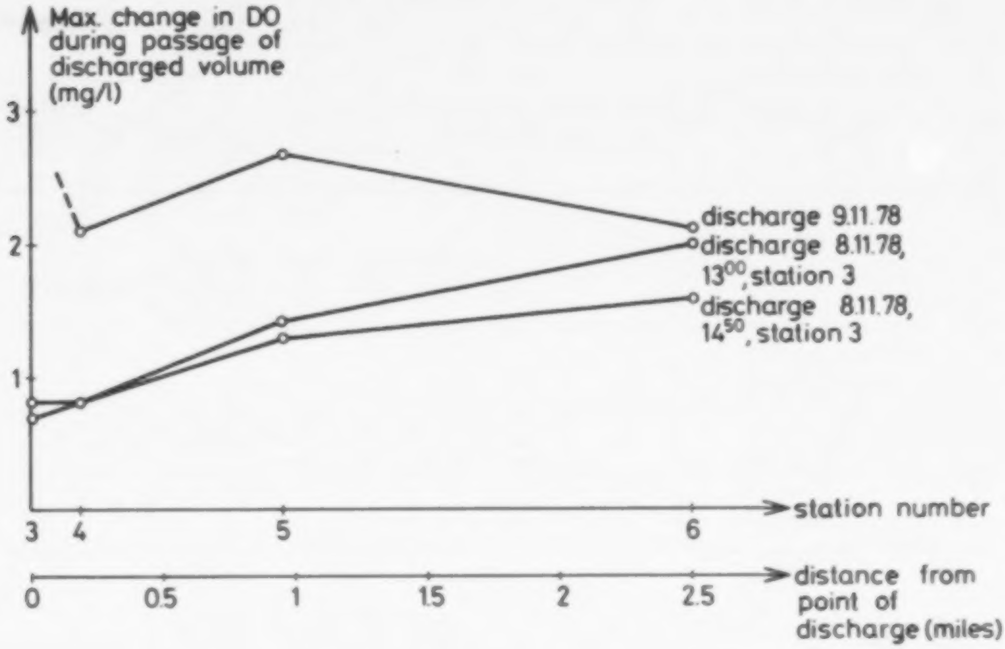


Fig. 5: Maximum difference in DO-concentration between the actual value during passage of discharged watervolume and normal expected value if no discharge took place.

Based on measurements of 9.11.78 with a stream temperature of 9°C the following parameters were calculated during passage of the polluted watervolume:

Rate of stream deoxygenation $K_1 = -0.15 \text{ day}^{-1}$

Rate of total BOD-removal in the stream $K_r = -0.55 \text{ day}^{-1}$

In smaller streams with values of K_1 and K_r as mentioned above and at the actual reaeration ($K_2(20) \approx 7 \text{ day}^{-1}$) it does not seem possible to maintain a difference greater than about 2 mg/ℓ between actual DO-concentration in the stream during a discharge event as simulated and the normal value if no discharge took place. If greater change in DO-concentration in such situations can be avoided immediately after mixing combined sewer overflow with stream water this seems to be a very important factor for maintaining sufficient DO in the stream.

DELAYED EFFECT IN THE STREAM

As mentioned before the delayed effect on the DO-concentration in the stream must be caused by adsorption and sedimentation of organic matter on the bottom due to the fact that removal of BOD from the watervolume exceeds actual degradation during storm events. On fig. 4 this delayed effect can be observed. At station 3 thus after 4 p.m., at the stations 4, 5 and 6 correspondingly displaced time of travel 15, 60, and 130 minutes. Measurements with tracer (rhodamin B) indicate that the observed effects on DO-concentration cannot be due to delay of polluted water in the dead-zone volume of the stream. Dependent of the actual possibilities for adsorption, sedimentation, desorption and resuspension the observed delayed effect on the DO-concentration will, with respect to magnitude and extension in time, be changed from one discharge event to another. The hydraulic conditions in the stream may be important for this effect. The delayed effect seen on fig. 4 shows, however, what was observed several times. It is thus typically that the delayed effect had a duration of about 24 hours, and that the situation then was near the steady-state situation.

The calculated total respiration (R_{tot}) of the stream immediately before discharge on 8.11.78 will, due to decreased influence by the sewage treatment plant, be reduced downstream (fig. 6). If values of R_{tot} are calculated immediately after discharge on 8.11.78 and 9.11.78 an increased relative importance is observed at the most distant stations (fig. 6).

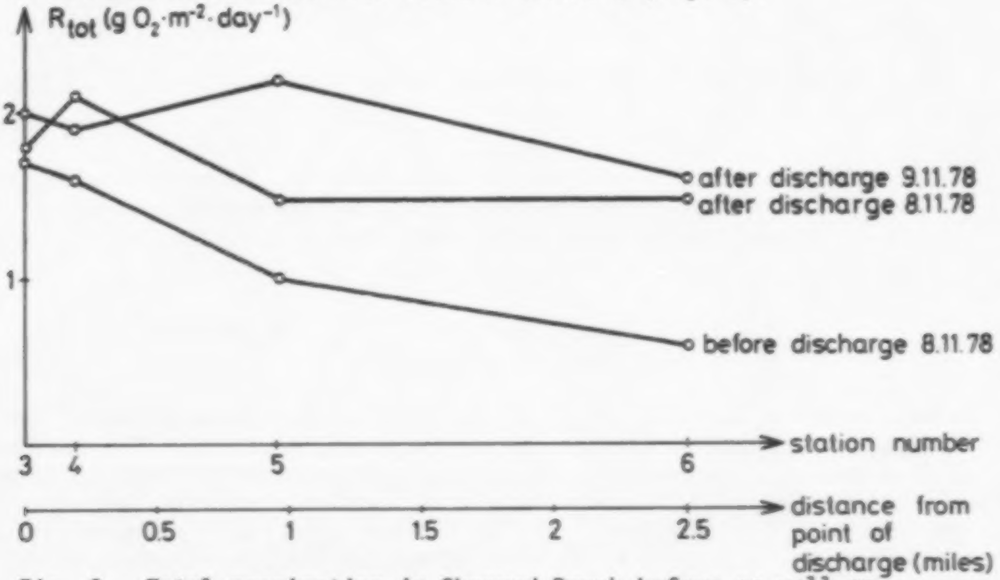


Fig. 6: Total respiration in Skravad Brook before as well as after discharge on 8.11.78 - 9.11.78.

In spite of the fact that the measured changes in the respiration of the stream are relatively large ($\sim$ 100%) the absolute values are, however, of insignificant influence on the DO-concentration in the described case. The oxygen consumption by degradation of the adsorbed colloidal and settled particulate matter is thus spread over such a long period that the effect on DO-concentration is reduced.

As mentioned, the actual conditions for adsorption and sedimentation of discharged organic matter will influence the delayed effect on the DO-concentration. To throw light on this there were set up mass balances for the section between stations 3 and 5. The calculations were made for SS and COD_{part}, the latter defined as the difference between the COD of a non-filtered and a filtered sample. Fig. 7 shows the processes influencing the SS mass balance.

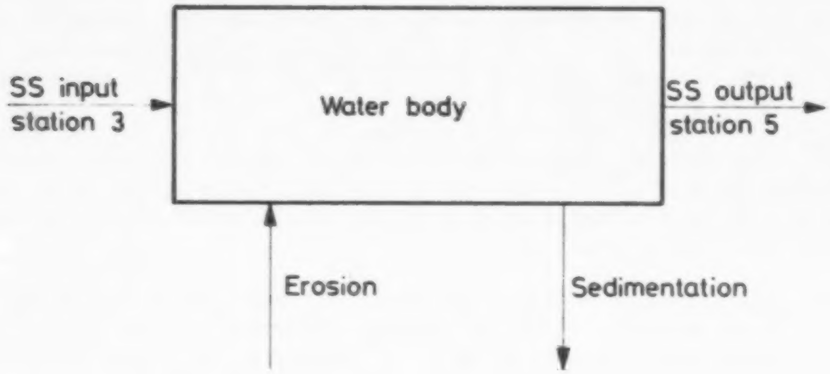


Fig. 7: Mass balance for SS for the section between station 3 and 5.

Based on measurements during the period of discharge on 9.11.78 the results are as follows:

For SS:

Erod - sed = Input - Output = 12 kg SS/80 min.
($\sim$35% of mass input at station 3).

For COD_{part}:

Erod - Sed = Input - Output = 1.1 kg COD/80 min.
($\sim$ 4% of mass input at station 3).

These results indicate that, due to the increase in water level owing to the hydraulic conditions, erosion takes place in the stream. The substance which is eroded has, however, in comparison with what is discharged, a much lower value of COD (10-50 mgO_2/g dry matter contrary to about 1000 mgO_2/g dry matter). Only unimportant quantities of particulate organic matter are thus added to the bottom. The delayed effect on the DO-concentration is thus likely to be due to adsorbed colloidal organic matter at the bottom.

RESULTS FROM REAL DISCHARGES

As mentioned, the investigations were mainly concerned with the impact of simulated discharges. However, current measurements of a few parameters took place. It was thus demonstrated that the actual sewerage system normally does not cause great changes in the DO-concentration in the stream immediately after mixing combined sewer overflows and water from the stream. Results obtained at station 3 for 30 of a total of 66 discharges in the period 30.6.78 - 20.11.78 are shown on fig. 8.

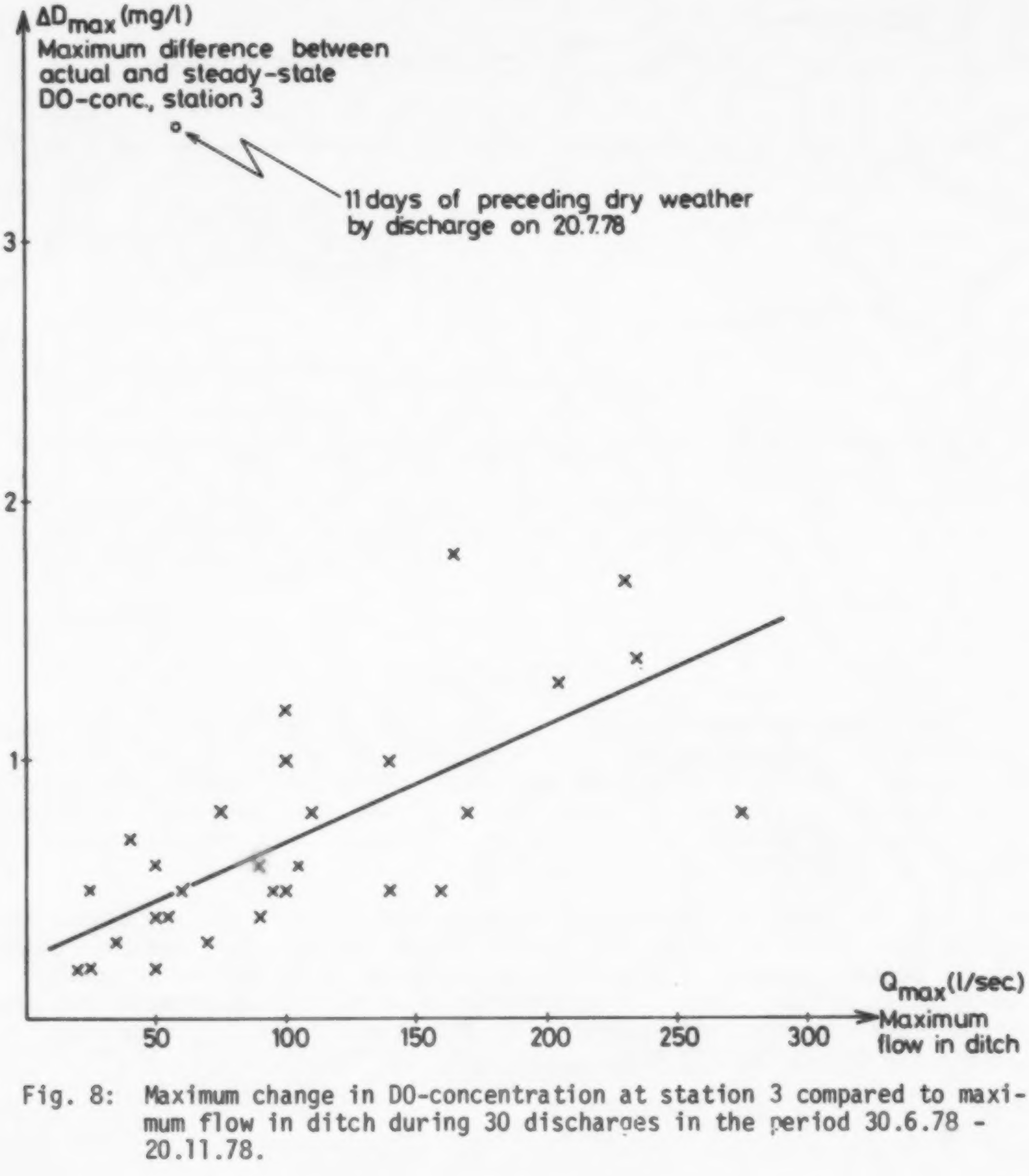


Fig. 8: Maximum change in DO-concentration at station 3 compared to maximum flow in ditch during 30 discharges in the period 30.6.78 - 20.11.78.

An arbitrary example of these real discharges that took place on 5.10.78 at 10:00-16:00 with maximum flow 170 ℓ/sec. in the ditch at 13:20 is shown on fig. 9. As found during simulated discharges an immediate effect as well as a delayed effect on DO-concentration is observed. This fact does not indicate any different impact between real and simulated combined sewer overflows on the DO-concentration of the stream.

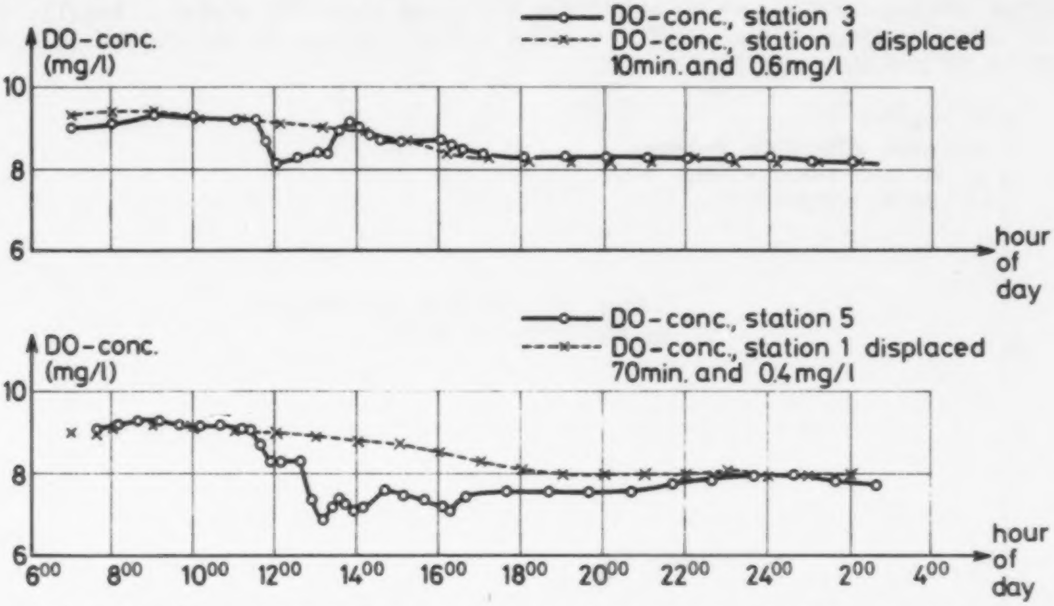


Fig. 9: DO-concentrations at the stations 3 and 5 before, during and after discharge on 5.10.78 compared to expected DO-concentration if no discharge took place (displaced curve from station 1).

CONCLUSIONS

The investigations show that combined sewer overflows to a small stream result in an immediate as well as a delayed impact on the DO-concentration in the stream. Due to this fact the degradation of organic matter loading on to the stream is spread over a longer period of time than just the time length of discharge of polluted overflow water. This result is obtained from real as well as simulated overflow situations.

The importance of this effect to water quality will largely depend on actual possibilities for degradation of BOD in the waterbody and absorption at the bottom compared to adsorption of colloidal and settling of organic particulate matter. Actual DO-concentration in the stream after mixing combined sewer overflows with stream water and actual reaeration rate will affect the stream quality too.

REFERENCES

1. Schaarup-Jensen, K., T. Hvitved-Jacobsen and E. Mortensen: Recipienters reaktion pa regnvandsafledning fra et fælleskloakeret byomrade, april 1979.

2. Harremoes, P.: Betydningen af forureningen fra regnafstromning for valg af urbane aflobssystemer. - En oversigt over nordisk litteratur og vurdering af status. Trettonde nordiska symposiet om vattenforskning, Roros, 2.5-5.5 1977. Nordforsk publikation 1977:2.

3. Johansen, L.: Afledning af regnvand, may 1974.

4. Horner, R.W., L.B. Wood and L.R. Wroe: London's stormwater problem, Water Pollution Control Federation, Journal, vol. 49, No. 1, 1977.

5. Pitt, R. and R. Field: Water-Quality effects from urban runoff, American Water Works Association, Journal, vol. 69, 1977.

6. Smith, R. and R.G. Eilers: Effect of Stormwater on Stream Dissolved Oxygen, Journal of the Environmental Engineering Division, Proceedings of the American Society of Civil Engineers, vol. 104, No. EE4, 1978.

7. Streeter, H.W. and E.B. Phelps: A Study of the Pollution and Natural Purification of the Ohio River, US Public Health Service, 1925.

8. Harremoes, P.: Udledning af organisk stof i vandlob, Vand 4, 1979.

9. Hvitved-Jacobsen, T.: Et vandlobs iltforhold under pavirkning af regnvandsafledning fra et fælleskloakeret byomrade, Vand 4, 1979.

6

Fifth Session

IMPACTS ON ESTUARIES

Moderator: Raymond P. Canale
University of Michigan
Ann Arbor, Michigan

IMPACTS OF INTERMITTENT URBAN DISCHARGES ON RECEIVING WATERS

John L. Mancini
Manhattan College
Bronx, New York

Abstract

The paper will present an illustration of the impacts of CSO discharges on bathing beach areas of New York City. The illustration will also define the treatment required and the costs for controlling water quality. A unique statistical analysis used in the project will be discussed.

The proposed paper will be divided into two segments.

Segment #1 - will discuss an analysis of coliform distribution in New York Harbor. The paper will present calculated and observed time variable coliform distributions over several storm events for the major water bodies that make up New York Harbor. Data and calculations will be presented for different size rainfalls and for different Hudson River flows. This segment of the paper will conclude with economic data on the cost of controlling coliform levels at selected beach locations in New York Harbor. Therefore, loads, fate and effects of storm water associated coliform distributions will be discussed.

Segment #2 - will address the fate and effect of intermittent discharges of toxics. In particular, information from data in the Trinity River, will be employed to calculate partition coefficients and equivalent removal rates for Cd, Cu, and other metals. Comparable calculations for the equivalent removal of Lindane, DDE, DDT and several other organic toxics will also be provided. This will be a discussion of fate of toxics.

The effects part of this segment will employ dose response data for Cd to illustrate calculations of mortality of an organism due to a storm load. The effect part of the paper will present a procedure for calculations. In situ data are not available to test the calculation procedure. The issues of concern in rational evaluation of effects will be identified in a quantitative manner.

Introduction

This paper deals with three interrelated considerations for defining the impacts of intermittent discharges on receiving waters. Initially a classical approach employing non-steady state water quality modeling for coliform in New York Harbor is discussed. The problem setting is a complex one due to the number of sources, the stratified nature of the estuary and the general scale of the analysis. It is, however, an illustration of the application of well defined technologies for the analysis of the impacts of intermittent discharges. The second part of the presentation addresses a technique which can be considered for defining a wet weather equivalent to the 7 day - 10 year low flow criteria used in many water quality standards. This technique may also be considered as one method of providing information on the probable setting for local and regional water quality problems caused by intermittent discharges.

The third section of the presentation illustrates one method of employing existing data on biological effects to begin addressing the questions associated with numerical wet weather criteria which could ultimately be considered in the context of wet weather standards.

The second and third segments of the presentation deal with the very initial results of an on-going research effort and are therefore preliminary in nature. It is hoped that a technical dialogue on wet weather water quality criteria will be stimulated.

New York Harbor Analysis of Coliform

Figure 1 is a map of the New York metropolitan area. The location of bathing beaches at which elevated coliform levels have been observed is shown on the map. One objective of the recent New York 208 Study was to define the treatment required to protect these beaches during the swimming season(1). Figure 2 illustrates the segmentation of the water quality model and the load generation model. The loadings to the system consisted of treatment plant discharge, leakage from regulators and discharges from both combined and separate sewered areas. The water quality model was a tidally averaged non-steady state calculation which included vertical segmentation and associated analyses to represent the circulation in this stratified estuary(2). The water quality model was compared to observed chloride and coliform data collected under seven conditions as illustrated in Table 1(3). The rainfall varied from 9.5 inches to .3 inches while the peak Hudson River flow ranged from 115,000 cfs to 10,000 cfs.

Figure 3 illustrates the comparisons of observed and calculated chloride for a period when the flow ranged between 21,000 and 58,000 cfs. Figures 4 and 5 present comparisons of calculated and observed coliform levels for two surveys considering the Hudson River and the East River. The comparisons of calculated and observed data were considered adequate to provide a basis for examining treatment requirements. In general the results of the analysis indicated that discharges within 4 to 6 miles of the beach required control and treatment; while those further away from the areas to be protected had smaller effects on coliform levels at the beach and treatment was not required to meet existing coliform standards. These findings provide an opportunity to stage capital and operating expenditures over time obtaining benefits in a sequential manner directly associated with expenditures. The cost of

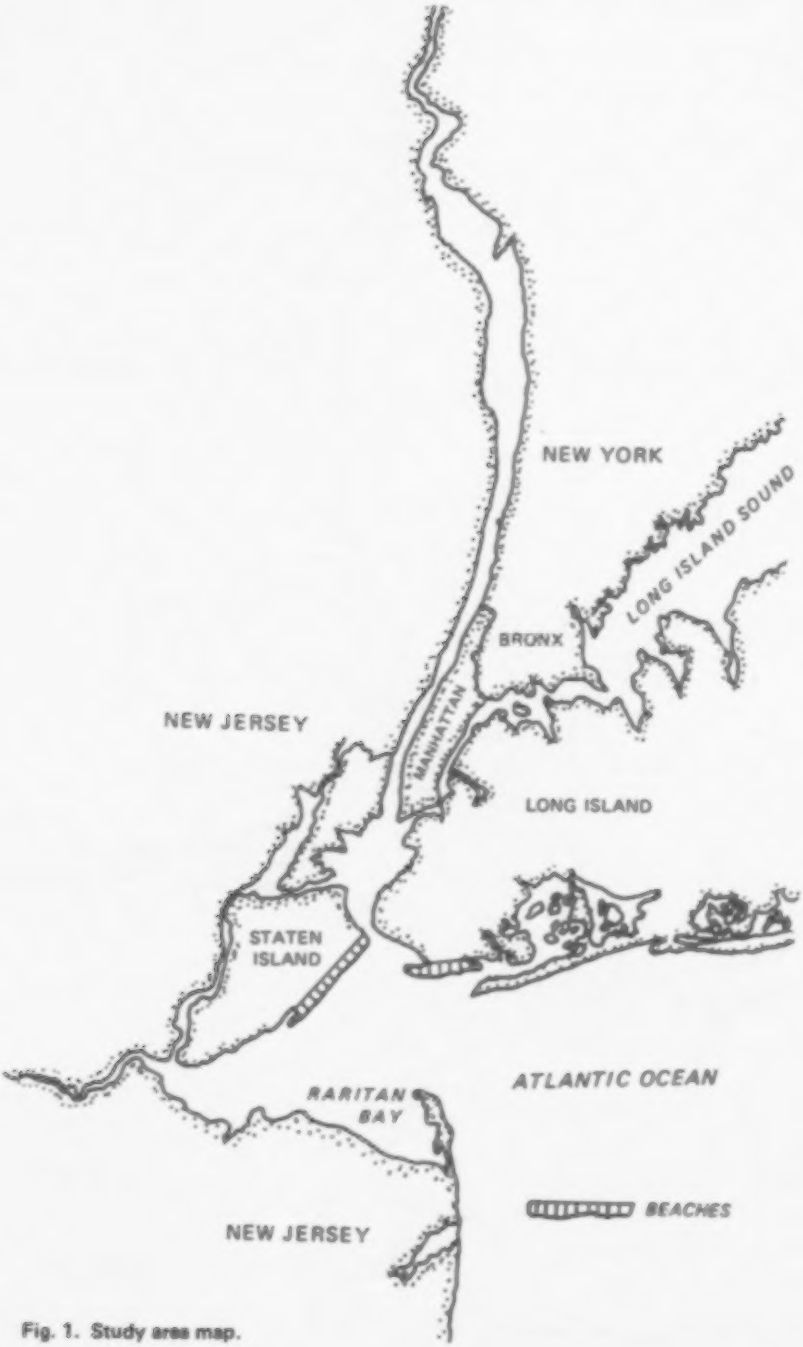


Fig. 1. Study area map.

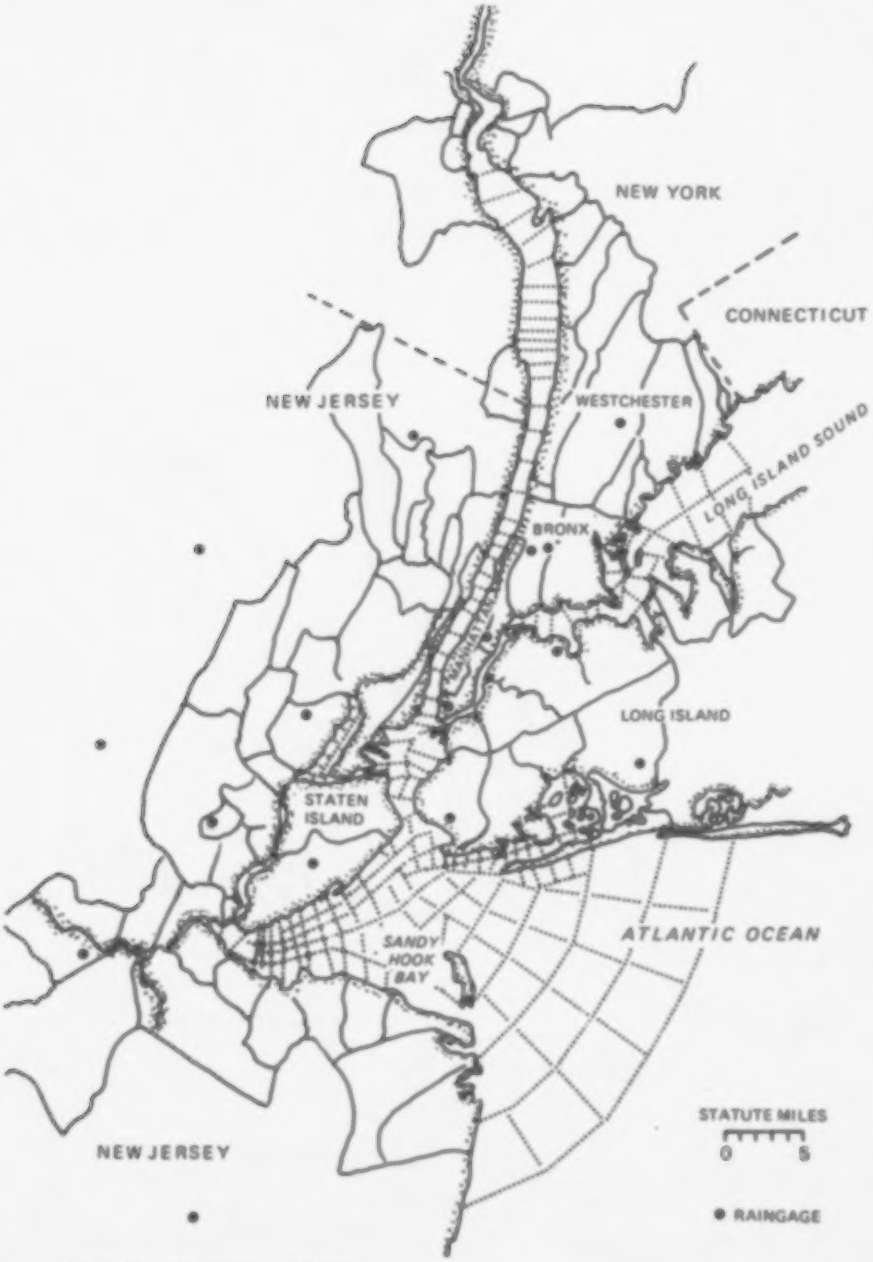


Fig. 2. Areawide rainfall-runoff model.

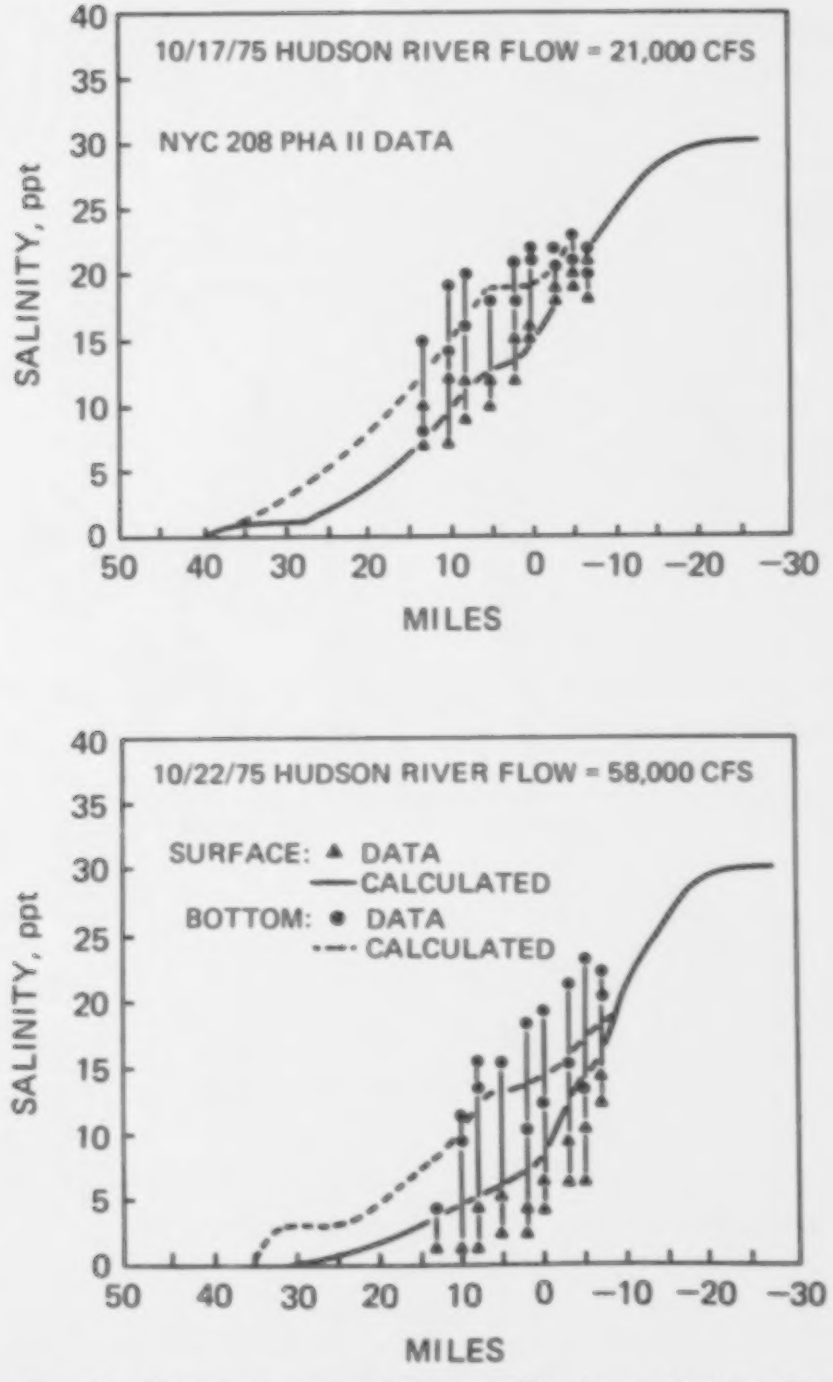


Fig. 3. Illustration of salinity verifications, Hudson River transect.

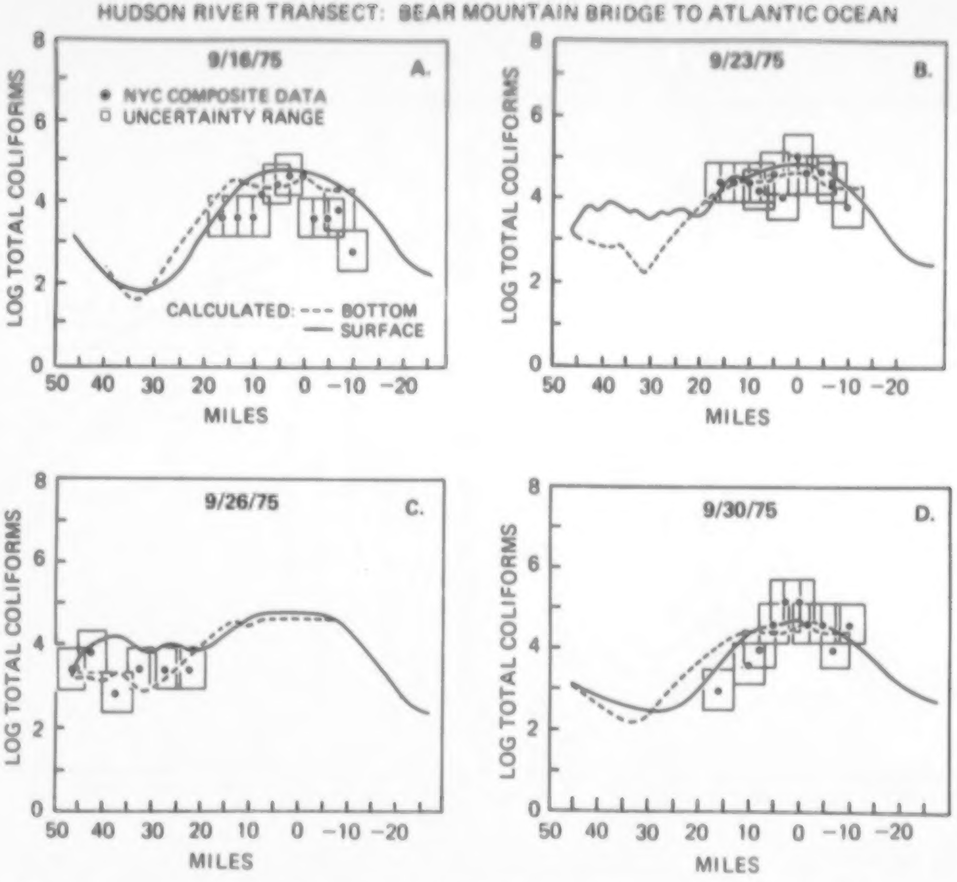
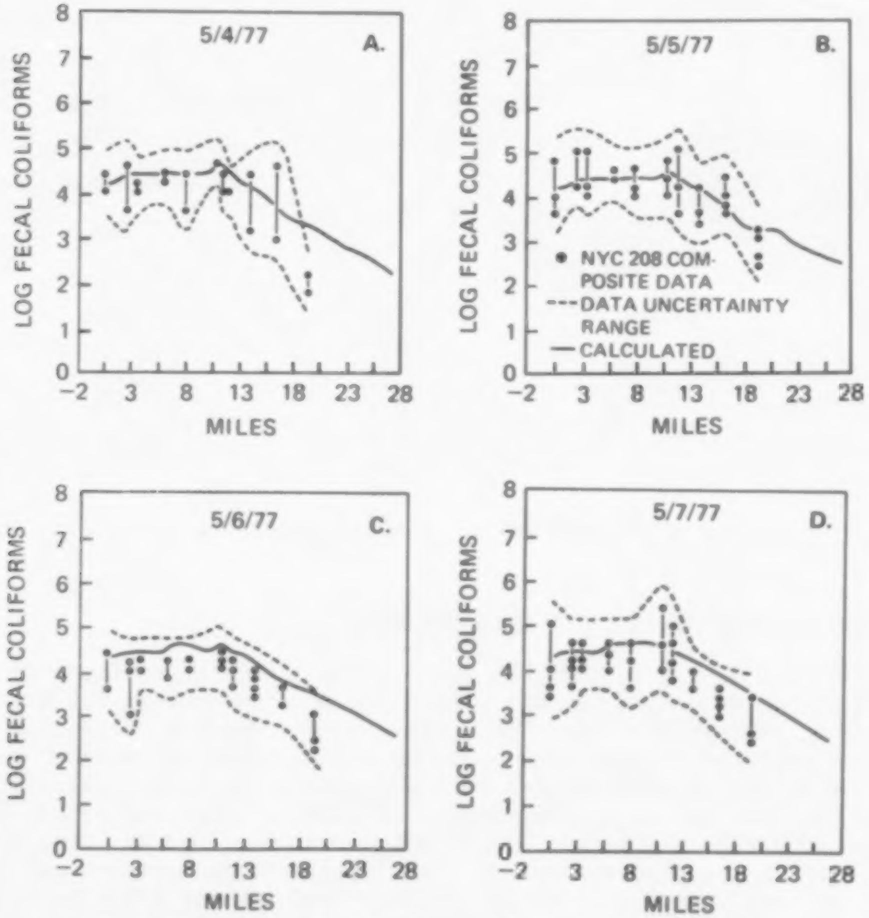

HUDSON RIVER TRANSECT: BEAR MOUNTAIN BRIDGE TO ATLANTIC OCEAN

Fig. 4. Illustration of total Coliform verifications, Hudson River transect, September, 1975.



EAST RIVER TRANSECT, UPPER BAY TO LONG ISLAND SOUND

Fig. 5. Illustration of fecal Coliform verifications, East River transect.

controls, even on this scale, are substantial and could represent 200 million dollars. The economic feasibility of the individual components is still under discussion.

In summary, a well defined technology base exists to analyze receiving water impacts associated with intermittent discharges from combined sewer overflows and urban runoff.

TABLE 1

VERIFICATION CONDITIONS FOR THE COLIFORM ANALYSIS IN NEW YORK HARBOR

Condition	Rainfall* (ins)	Peak Hudson River Flow (cfs)
1	9.5	71,000
2	2.6	81,000
3	2.2	25,000
4	.8	115,000
5	1.4	24,000
6	0.3	10,000
7	5.1	11,000

* Total average rainfall from all operating gauges.

Wet Weather Frequency of Occurrence Analysis

Treatment of all runoff from urban areas would be prohibitively expensive and the water quality benefits have not been assessed. One method of beginning to address this issue is to examine wet weather criteria which could eventually be considered for use in establishing wet weather standards. One of the primary issues relates to the degree of environmental protection which is required or can be supported by available money and technology. The degree of environmental protection is often characterized in terms of recurrence frequency and a concentration associated with some biological or health effect. As an illustration, water quality standards are often written in terms which state that the dissolved oxygen shall be above 5 mg/ℓ at flows equal to or greater than that associated with the 7 day - 10 year low flow. This low flow has a value which can be determined by analysis of the stream flow records at a location. Even if records do not exist at a given location there are techniques available to estimate this flow based on drainage area and records for similar nearby streams. There is a need to define a comparable frequency related parameter for wet weather events.

One parameter which could be considered is related to the joint occurence of stream flow and rainfall. Historical records are available for stream flow and for rainfall in most regions of the country. Methods for extrapolation to specific sites are either available or could be developed. As an illustration of the approach, daily stream flow records from Williams Creek in Austin, Texas were paired with daily rainfall data from this city for the period October 1, 1976 to September 30, 1977. The number of occurences of both rainfall and flow were determined and the results are presented in Table 2. It may be seen from the table that most of the rainfall events were associated with higher stream flows. However, a number of rainfalls occurred during low flow periods. {One method of determining the relative impact of runoff loads at various combinations of rainfall and stream flow is to calculate the dilution available in the stream.} This can be done by defining the runoff coefficient and the extent (size) of the urban area. Figure 6 contains an idealized illustration of a site. The runoff volume is defined by the product of the rainfall intensity, urban drainage area, and runoff coefficient. This volume is divided by a storm duration to yield a runoff flow. In the example, daily rainfall data was used and the duration is assigned consistent with this time period. It should be noted that hourly rainfall data is usually available at class A stations and could be used in an analysis. Assuming that the runoff coefficient is 0.2 and that the relative drainage area of the stream was 100 times that of the urban area, the calculated dilution factor in the stream is presented in Table 3 for urban runoff from a separately sewered area. The dilution factor is defined by:

$$\text{Dilution Factor} = \frac{\text{Runoff Flow}}{\text{Total Estimated Downstream Stream Flow}}$$

This factor may be multiplied by the concentration of a contaminant in the runoff to yield an approximation of the maximum instream concentration. The total estimated downstream flow has been defined by the sum of the runoff flow and the stream flow.

The event with the lowest dilution (i.e. highest dilution factor) can be considered the most critical for many parameters. Dissolved oxygen is one parameter where this may not be true. The combinations of rainfall and stream flow can be ranked based on the dilution factor. In the illustration, the most critical condition occurs at a rainfall in the range of 0.1 to 0.2 inches which was associated with a stream flow of below 0.1 cfs. Under this condition the maximum instream contaminant concentration would be approximately 83 percent of the concentration in the runoff. If desired, the ranking can be weighted by the probability of occurrence related to the total period analyzed (i.e. 365 days) or by the number of days of rain (in this case, 88). These procedures could be employed to define frequency in terms of sequential events over one or several seasons or years. An alternate approach could be considered which uses an extreme value analysis technique similar to that employed to define the 7 day - 10 year low flow used in current stream standards. This procedure employs the most critical event for each year of record and the frequency distribution defined by these events is examined to determine the basis for environmental protection.

Table 4 has been developed considering the situation presented in Figure 6 and assuming that the intermittent discharge is from a combined sewer

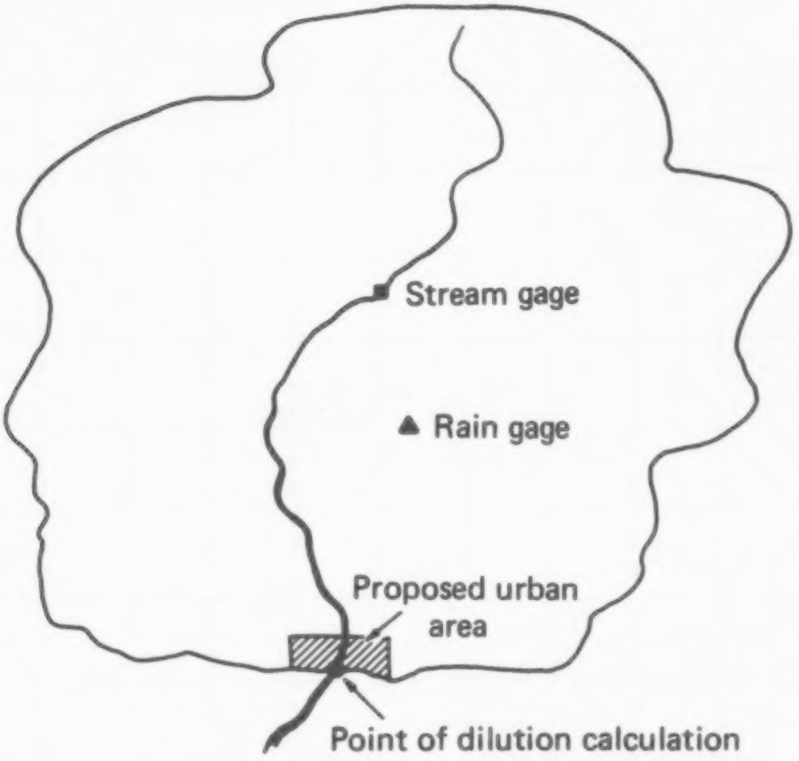
TABLE 2

NUMBER OF EVENTS OF JOINT STREAM FLOW AND RAINFALL
WILLIAMS CREEK, AUSTIN, TEXAS

October 1, 1976 to September 30, 1977

Stream Flow (cfs) \ Daily Rainfall (ins)	0(1) – .005	.005 – .05	.05 – .1	.1 – .2	.2 – .3	.3 – .4	.4 – .6	.6 – .8	.8 – 1.1	1.1 – 1.5	1.5 – 5
0 – .01	0										
.01 – .05	5										
.05 – .1	22	2	1		1						
.1 – .2	24										
.2 – .4	28	2		1							
.4 – .8	9			1							
.8 – 1.5	18	3									
1.5 – 3	45	3	1	5	1	1					
3 – 6	84	14	4	4	3		4	1		1	
6 – 15	32	6	1	2		2		3	1		
15 – 300	10	3		1			2	3	6	3	2

NOTE: 1) included trace rainfalls and no rain
2) 88 Rain days
3) 365 days of record



Note: Stream flow translated on a drainage area basis.

Fig. 6. Site used in the example calculation.

TABLE 3

CALCULATED INSTREAM DILUTION FOR URBAN RUNOFF EVENTS(1)(2)(3)

Daily Rainfall (ins)

Stream Flow (cfs)	0–.005	.005–.05	.05–.1	.1–.2	.2–.3	.3–.4	.4–.6	.6–.8	.8–1.1	1.1–1.5	1.5–5
0–.01	(4) -										
.01–.05	-										
.05–.1	-	.35	.60		.83						
.1–.2	-										
.2–.4	-	.12		.43							
.4–.8	-			.27							
.8–1.5	-	.03									
1.5–3	-	.02	.05	.09	.14	.19					
3–6	-	.01	.02	.05	.08		.14	.19		.30	
6–15	-	0	.01	.02		.05		.09	.16		
15–300	-	0		0			0	0	0	.01	.03

NOTES:
1) Stream Drainage Area/Urban Area = 100:1
2) Runoff Vol/Rain Vol = .2
3) Runoff duration = 24 Hrs.
4) No overflows

TABLE 4

CALCULATED INSTREAM DILUTION FOR CSO EVENTS(1)(2)(3)(5)

Daily Rainfall (ins)

Stream Flow (cfs)	0	.005	.05	.1	.2	.3	.4	.6	.8	1.1	1.5	5
0	(4)											
.01	-											
.05	-	-	-		.52							
.1	-											
.2	-	-		-								
.4	-			-								
.8	-	-										
1.5	-	-	-	-	.03	.14						
3	-	-	-	-		.08						
6	-	-	-	-		.03		.12	.17			
15	-	-	-	-		.01	.01	.01	.01	.02	.05	
300												

NOTES:
1) Stream Drainage Area/Urban Area = 100:1
2) Runoff Vol/rain Vol = .4
3) Runoff Duration = 24 Hrs.
4) No overflows
5) Interceptor Capacity = 6 cfs/mi^2
6) Number of overflows = 34

system with an excess interceptor capacity of 6 cfs/sq. mi., a runoff coefficient of .4 and the ratio of stream to urban drainage area of 100:1. The dilution factor in the stream tends to decrease due to the reduction in runoff associated with the interceptor. In addition, the number of days in which overflows occur are reduced by the interceptor from 88 to 34. The concentration of most constituents is substantially higher in combined overflows as compared to urban runoff; therefore, the water quality impacts would be larger.

In summary, one method for defining critical wet weather events has been identified and can be considered for use in developing frequency information which could provide a part of the basis for wet weather water quality criteria. The approach illustrated is in the early stages of development and employed a large number of simplifying assumptions. In its most generalized form the approach could consider hourly rainfall and/or stream flow records, variable runoff coefficients, and variable constituent concentrations as a function of rainfall intensity or volume. An additional use which can be made of the proposed approach is as a screening methodology for identifying the relative size of river and urban areas in which water quality problems could be anticipated. The critical relative size would vary regionally due to climate and also be related to the intensity of urbanization. The approach has not been tested and application should not be considered until adequate development and site specific testing has been carried out.

Wet Weather Biological Impacts

As indicated previously the second part of wet weather criteria deals with the biological or health effects of in-stream concentrations of contaminants. In the context of wet weather phenomena, it is necessary to consider the time of exposure in addition to the concentration of the contaminant. This is due to the transient nature of the intermittent loads from storm runoff events. As an illustration of one possible approach to evaluation of biological impacts the following example has been developed.

Figure 7 contains data on the percent survival of the snail (*Physa integra*) as a function of time for several concentrations of cadmium. These data are replotted on Figure 8 employing coordinates which can be employed to develop equation (1) relating percent survival to exposure time and cadmium concentration.

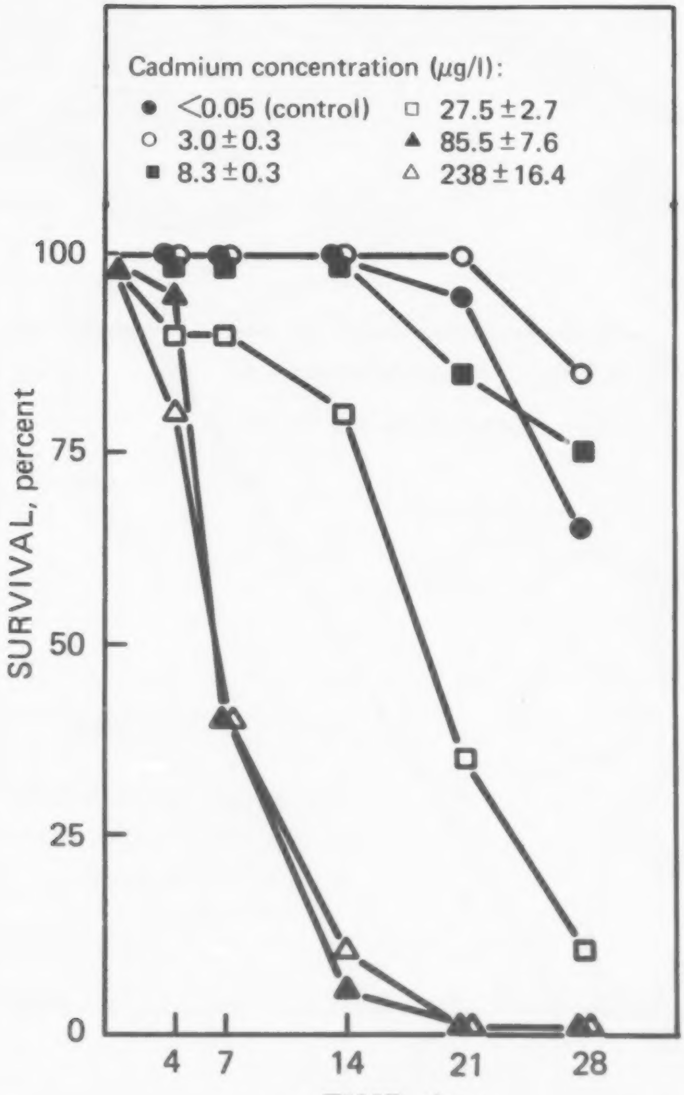
$$K = 0.3\ t\ c^{0.6} \qquad (1)$$

where:

K = % kill
t = exposure time (days)
c = cadmium concentration (μg/ℓ)

Rewriting equation (1) in terms of population and differentiating with respect to time yields:

$$\frac{dP_t}{dt} = -\frac{.3C^{.6}}{100} \times P_o \qquad (2)$$



From Spehar, R. L., Anderson, R. L., and Fiaudt, J. T., *Environ. Pollut.* 15, 1978, 195.

Fig. 7. Percent survival of *Physa integra* (snail) after 28 days of exposure to cadmium

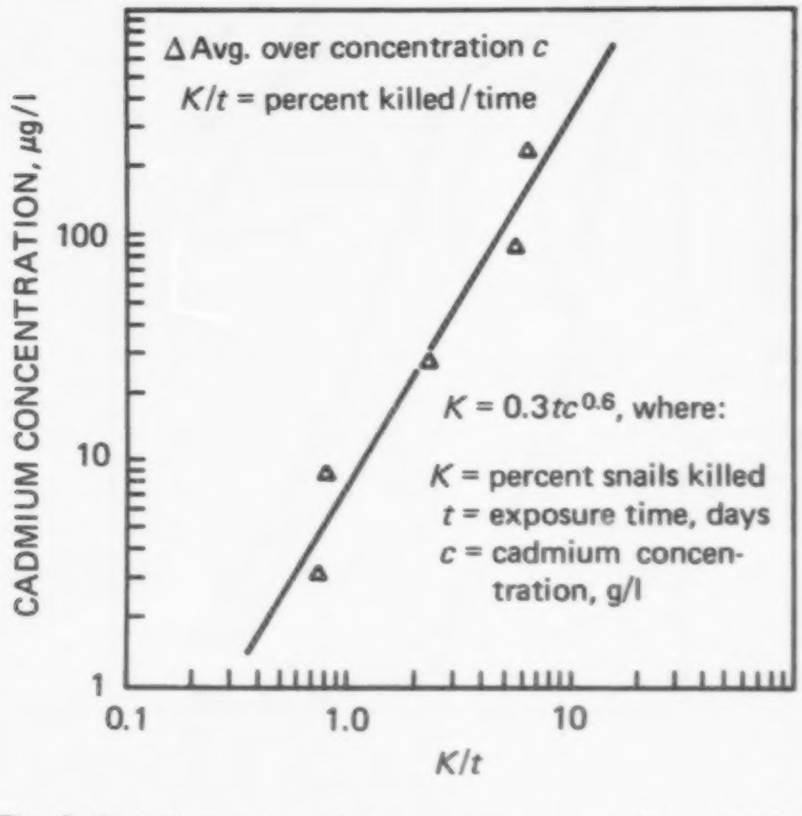


Fig. 8. Cadmium concentration vs. average snail mortality/time.

where: P_o = initial snail population

P_t = snail population at time "t"

The data employed was from a series of mortality experiments conducted at different concentrations of cadmium. The cadmium concentrations were maintained at a constant level in each individual mortality experiment. The expression in equation (2) defines the rate of change of the snail population as a function of the time invariant cadmium concentration.

A storm water discharge into a completely mixed lake can be considered. In order to minimize the mathematical complexities of this illustration zero ourflow from the lake was assumed. The time history of cadmium concentration in the lake is defined by equation (3) assuming cadmium is removed by sorption and settling at a rate "$\bar{k}$"(5).

$$C = Co\ e^{-\bar{k}.t} \tag{3}$$

where: C = cadmium concentration at time "t" (μg/ℓ)

C_o = initial cadmium concentration (μg/ℓ)

$\bar{k}$ = equivalent rate of removal (1/Day)

t = time (Days)

Assuming that a measure of the rate of change of the population in experiments run at constant but different concentrations is equal to the rate of change of the population when the concentration is varying with time [illegible] allows substitution of equation (3) into equation (2) and integration yields:

for P_t = Po @ t = o and C = Co @ t = o

$$P_t = Po(1 + \frac{C_o^{0.6}}{\bar{k}x200}(e^{-\bar{k}\ x\ .6\ x\ t} - 1)) \tag{4}$$

Equation (4) relates the population of snails to the cadmium concentration and time after a storm event in a simplified lake system. Calculations are presented in Tables 5 and 6 using equation (4). In Table 5 it is assumed that a small lake receives a large urban load generated by three different rainfall events. The calculations in Table 6 are for the same rainfall events considering a lake which is five times as large. Information on population reductions can be combined with estimates of the frequency of rainfall occurence to provide a basis for criteria to be employed in evaluation of wet weather standards for different types and sizes of water bodies.

In any generalization of the above illustration the initial cadmium concentration "Co" would reflect the size of the urban drainage area, land use, size of the rainfall, antecedent conditions and other features of the system. In addition, the value of initial concentration "Co" would reflect the initial dilution for various sizes and types of water bodies and could

TABLE 5

```
 INIT. POPULATION= 500 NUMBERS
 INIT. CONC. = 20 PPB
 CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS        POPULATION       %KILLED          CHEM. CONC.
 0                500              0                20
 5                460.901          7.81975          12.1306
 10               431.936          13.6128          7.35759
 15               410.478          17.9043          4.4626
 20               394.582          21.0836          2.70671
 25               382.806          23.4389          1.6417
 30               374.082          25.1837          .995742
 35               367.619          26.4763          .603948
 40               362.831          27.4338          .366313
 45               359.284          28.1432          .22218
 50               356.656          28.6688          .134759

 INIT. POPULATION= 500 NUMBERS
 INIT. CONC. = 10 PPB
 CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS        POPULATION       %KILLED          CHEM. CONC.
 0                500              0                10
 5                474.205          5.15911          6.06531
 10               455.095          8.98107          3.67879
 15               440.938          11.8124          2.2313
 20               430.45           13.91            1.35335
 25               422.681          15.4639          .82085
 30               416.925          16.615           .497871
 35               412.661          17.4678          .301974
 40               409.502          18.0996          .183156
 45               407.162          18.5676          .11109
 50               405.428          18.9143          .0673795

 INIT. POPULATION= 500 NUMBERS
 INIT. CONC. = 3.33333 PPB
 CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS        POPULATION       %KILLED          CHEM. CONC.
 0                500              0                3.33333
 5                486.656          2.66871          2.02177
 10               476.771          4.64574          1.22626
 15               469.448          6.11036          .743767
 20               464.023          7.19539          .451118
 25               460.004          7.99918          .273617
 30               457.027          8.59465          .165957
 35               454.821          9.03579          .100658
 40               453.187          9.36259          .0610521
 45               451.977          9.60469          .03703
 50               451.08           9.78405          .0224598
```

TABLE 6

```
INIT. POPULATION= 2500 NUMBERS
INIT. CONC. = 4 PPB
CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS       POPULATION      %KILLED         CHEM. CONC.
 0               2500            0               4
 5               2425.57         2.97722         2.42612
 10              2370.43         5.18279         1.47152
 15              2329.58         6.81673         .892521
 20              2299.32         8.02717         .541341
 25              2276.9          8.9239          .32834
 30              2260.3          9.5882          .199148
 35              2247.99         10.0803         .12079
 40              2238.88         10.4449         .0732626
 45              2232.13         10.715          .044436
 50              2227.12         10.9151         .0269518

INIT. POPULATION= 2500 NUMBERS
INIT. CONC. = 2 PPB
CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS       POPULATION      %KILLED         CHEM. CONC.
 0               2500            0               2
 5               2450.89         1.96423         1.21306
 10              2414.52         3.41937         .735759
 15              2387.57         4.49736         .44626
 20              2367.6          5.29596         .270671
 25              2352.81         5.88758         .16417
 30              2341.85         6.32585         .0995742
 35              2333.74         6.65054         .0603948
 40              2327.72         6.89107         .0366313
 45              2323.27         7.06926         .022218
 50              2319.97         7.20127         .0134759

INIT. POPULATION= 2500 NUMBERS
INIT. CONC. = .666667 PPB
CHEM. REACTION RATE= .1 1/DAY
TIME/DAYS       POPULATION      %KILLED         CHEM. CONC.
 0               2500            0               .666667
 5               2474.6          1.01606         .404354
 10              2455.78         1.76878         .245253
 15              2441.84         2.32641         .148753
 20              2431.51         2.7395          .0902236
 25              2423.86         3.04554         .0547233
 30              2418.19         3.27225         .0331914
 35              2414            3.44021         .0201316
 40              2410.88         3.56463         .0122104
 45              2408.58         3.6568          7.406E-03
 50              2406.87         3.72509         4.49197E-03
```

include specific considerations such as river flow and dispersive mixing as appropriate. The type and size of the water body would also be included in the analysis and influence the relationship between the time "t" and the spatial location of calculated population reductions. In the current example snails were the organism examined. It was assumed that the snails were stationary in the lake. Organisms with different characteristics relative to the water column can be considered. As an example, plankton, which are transported with the storm pulse, could be analyzed as could fish which were confined to regions of the receiving water as the storm pulse passed. In general, equations (1) and (2) of the illustration would vary for different organisms and contaminants and could extend from classical water quality variables such as dissolved oxygen to toxics like cadmium and consider fish, plankton or bottom organisms.

The removal rate "$\bar{k}$" in the cadmium example would vary depending on the contaminant being studied and the type of water and/or water body. For classical water quality variables such as dissolved oxygen, suspended solids and coliform; conventional first order kinetics appear appropriate. For toxics it is possible to evaluate an equivalent first order removal rate (6). Once again the approach illustrated is in the early stages of development and significant additional effort and testing is required before application to site specific situations.

Conclusions

The impacts of intermittent discharges on receiving waters may be examined using an array of methodologies ranging from reasonably well tested approaches to those that are in the early stages of investigation. The latter approaches require further development and testing before they can be considered as practical tools. They do, however, hold some promise for providing a portion of the basis required to develop wet weather criteria which can be employed in the standards setting process.

References

1. Executive Summary - New York City 208, Hazen and Sawyer, 1978.
2. N.Y.C. 208 Task Report (PCP Task 314) "Seasonal Steady State Modeling", Hydroscience, Inc., 1978.
3. N.Y.C. 208 Task Report (PCP Task 334) "Time Variable Modeling", Hydroscience, Inc., 1978.
4. Spehar, R.L., Anderson, R.L. and Fiandt, J.T. Environ, Poll. 15, 1978, 195.
5. Manhattan College Summer Institute Notes - "Analysis of Natural Systems" 1979.
6. "Environmental Report Tenn. Coloney Res.", U.S. Army Corps of Engineers, Fort Worth Dist. Office, 1979.

THE RESPONSE OF GREAT LAKES ESTUARIES TO STORMWATER RUNOFF

Authors: John R. Adams
Stephen M. Yaksich, Ph.D
Army Corps of Engineers, Buffalo District
1776 Niagara Street
Buffalo, NY 14207

ABSTRACT

Glacial rebound of the northern shore of Lake Erie has resulted in drowned rivermouths of southern shore tributaries. Transport of material through these estuaries depends not only on river stage but also lake stage. This paper describes studies carried out in two Lake Erie tributaries during high and low flow conditions. It also examines the effect of localized urban runoff on one of the estuaries.

Three separate studies of water chemistry in the lower Maumee River, an estuarine river section, were conducted by the Toledo Metropolitan Council of Governments, the Buffalo District Corps of Engineers, and others, during 1974 and 1975. Base flow river conditions and the resultant estuary chemical variations were measured during the summer of 1974. Winter storm runoff effects were measured during January and February, 1975. Additional sampling and analysis was conducted during the summer of 1975, giving a fairly complete record of water chemistry variability.

Winter storm runoff was also measured in the estuary of the Cuyahoga River, with simultaneous river and estuary sampling over the hydrograph for selected pollutants.

This paper outlines the problems and complexities of chemical measurements of water quality in estuarine systems of the Great Lakes during varying hydrologic conditions. The studies in the Maumee River point out the importance of sampling program design, and the difficulty in evaluating the transport of pollutants through such estuarine systems. The summer studies were conducted by TMACOG as part of an evaluation of in situ water quality, as effected by a number of waste discharges from the Toledo area. The winter storm runoff measurements, made simultaneously at both an upstream riverine station and a downstream estuary station were designed to evaluate whether the mass transport being carried out into the basin was passing through the

estuary and out into the lake. The question of an increase or decrease in mass transport was also considered, to evaluate scouring and depositional mechanisms in the estuary.

The results of the Maumee estuary measurements were compared with the Cuyahoga estuary in Cleveland, where similar sampling of river and estuary transport was carried out for the same storm periods.

Introduction

Many tributaries to the Great Lakes have river mouths which can scientifically and legally be considered lacustrine estuaries. The fate of pollutants in and through these estuaries presents a problem for water resource planners. Transport of material through these estuaries depends not only on river stage but also lake stage. This paper describes studies carried out in two Lake Erie tributaries, the Maumee and Cuyahoga Rivers.

The studies considered include: the effects of water quality on the estuaries and the effects of the estuaries on water quality. That is, how the quality of runoff water from the agricultural watershed above the river mouths and the local urbanized areas effect the estuaries, and how the presence of an estuarine reach effects water quality in the estuary and the transport of water pollutants or nutrients to the water body beyond, in this case Lake Erie.

The effect of lake levels on the flow of river water through the estuary has a dramatic effect on the quality of water in the "river." It has previously been the habit of waste load allocation planners to treat these "river" segments in the same manner as any other free flowing segment. In actuality, the behavior of the estuaries is much more lake like.

The present studies have spent a large effort on refining estimates of loadings of pollutants and nutrients to Lake Erie. Part of this effort included streamflow and water quality parameter measurements in the estuaries and at upstream gaging stations during storm events. The purpose of these measurements was to provide estimates of the transport of storm runoff related pollution through the estuaries to Lake Erie.

Description of Sampling Programs

Three separate studies of water chemistry in the lower Maumee River were conducted by the Toledo Metropolitan Area Council of Governments (TMACOG), the Buffalo District Corps of Engineers, Lake Erie Wastewater Management Study (LEWMS), and USEPA, during 1974 and 1975. Base flow river conditions and the resultant estuary chemical variations were measured during the summer of 1974. Winter storm runoff effects were measured during January and February 1975. Additional sampling and analysis was conducted during the summer of 1975, giving a fairly complete record of water chemistry variability.

Figure 1 is a schematic diagram of the sampling locations utilizd in the summer water quality survey. Stations with numeric identification are TMACOG stations while those with alphabetic identification were utilized by Jones & Henry Engineers, Ltd. Samples collected at Stations 2, 5, 8, 11, 14, and 17 were analyzed for all the parameters listed in Table 1. The remainder of the numeric station samples received laboratory analysis only for fecal coliform organisms and chloride. All samples for laboratory analysis were taken at mid-depth with a Kemmerer or Van Doren Sampler, and were preserved immediately after collection by placing on ice. Laboratory analysis for nonconservative parameters was begun within 6 to 8 hours of the collection of the first sample. On a few occasions when immediate analysis was not possible, chemical preservation plus refrigeration was employed.

Table 1 - Parameter List: Maumee River Water Quality and Mass Transport Studies

---

Station Location
Date
Time
Conductivity 1/
Total Phosphorus 1/
Ortho Phosphorus 1/
Nitrate & Nitrite 1/
Ammonia 1/
Total Kjeldahl Nitrogen 2/
Total Solids 2/
Suspended Solids 1/
Total Organic Carbon 2/
SiO_2 1/
Chloride 1/
pH 3/
Extended Bod 3/
Fecal Coliforms 3/
Calcium 3/
Magnesium 3/
Potassium 3/
Sodium 3/
Iron 3/
Copper 3/
Manganese 3/
Disolved Oxygen 3/

1/ Analyzed in all studies

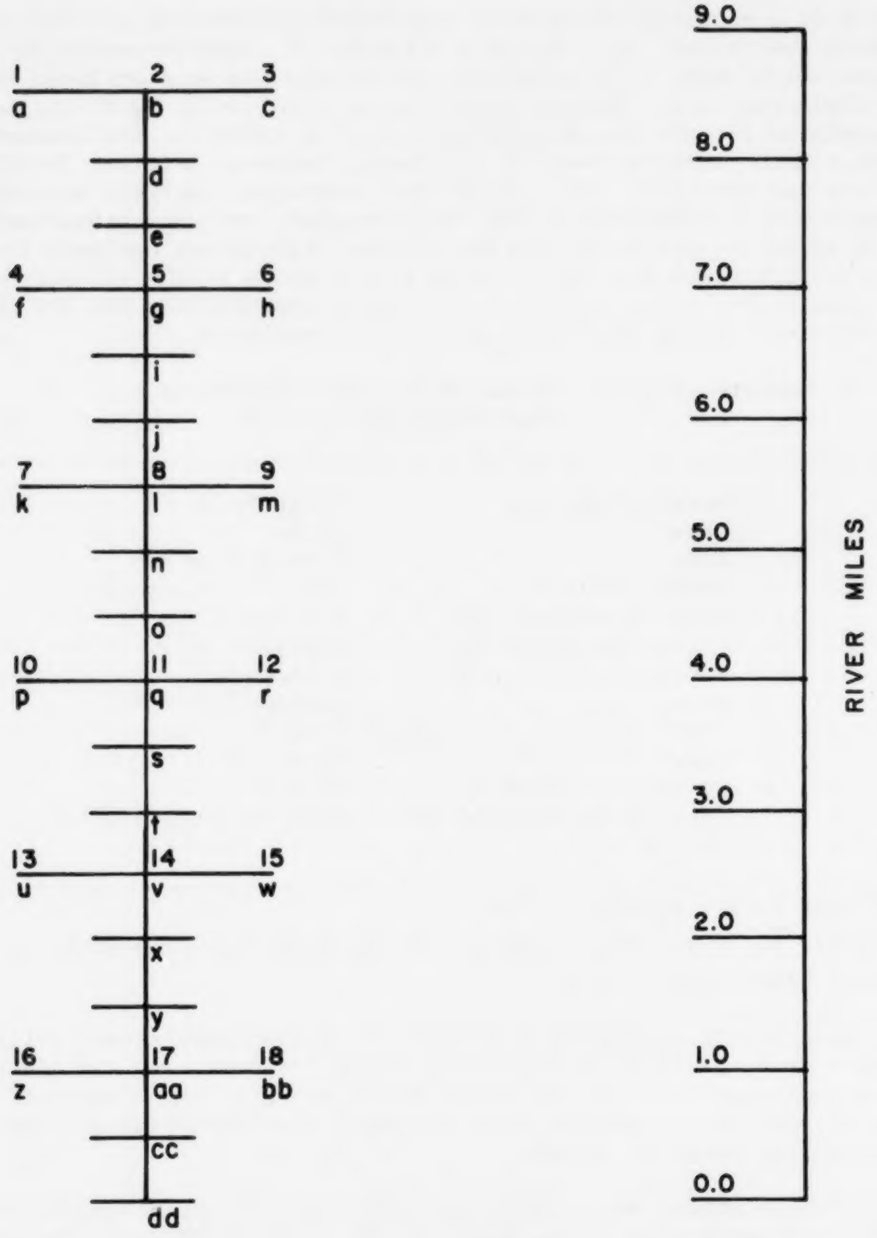
2/ Analyzed in Maumee Water Quality and Cuyahoga Mass Transport

3/ Maumee Water Quality only

At each of the alphabetic stations, field instruments were utilized to determine depth profiles of dissolved oxygen, temperature, and conductivity. The instantaneous depth of the water at the sampling point was also recorded to the nearest foot. Samples were collected once weekly at all stations throughout the sampling period.

Sampling stations were chosen on a grid pattern beginning at river mile (RM) 8.5, and extending to the mouth of the river at RM 0.0. The upstream station was chosen as being the probable furthest extent of the upstream influence of pollution contributed to the river in the heavily urbanized section of the river beginning around RM 6.5 and above all major point sources in the estuarine section of the river.

Independently from the water quality surveys conducted in the lower river during the summer of 1975, both the U. S. Army Corps of Engineers and TMACOG have engaged the Heidelberg College River Studies Laboratory in a study of



MAUMEE RIVER ESTUARY SAMPLING POINTS

FIGURE I

mass loading of pollution in the Maumee River Basin. In this study, a sample of the Maumee River has been taken every day between December 1974 and July 1977. Samples were collected by an automatic sampler which was located inside the pump house of the Bowling Green, Ohio water supply intake. The sampler collects one 500 ml sample every 6 hours. During periods of stable hydrograph, only one of the four daily samples was analyzed. Whenever a storm event occurred and the hydrograph rose significantly above base flow, all samples were analyzed to provide greater definition of the chemograph over the storm. Flows for this station are available from the USGS gaging station at Waterville.

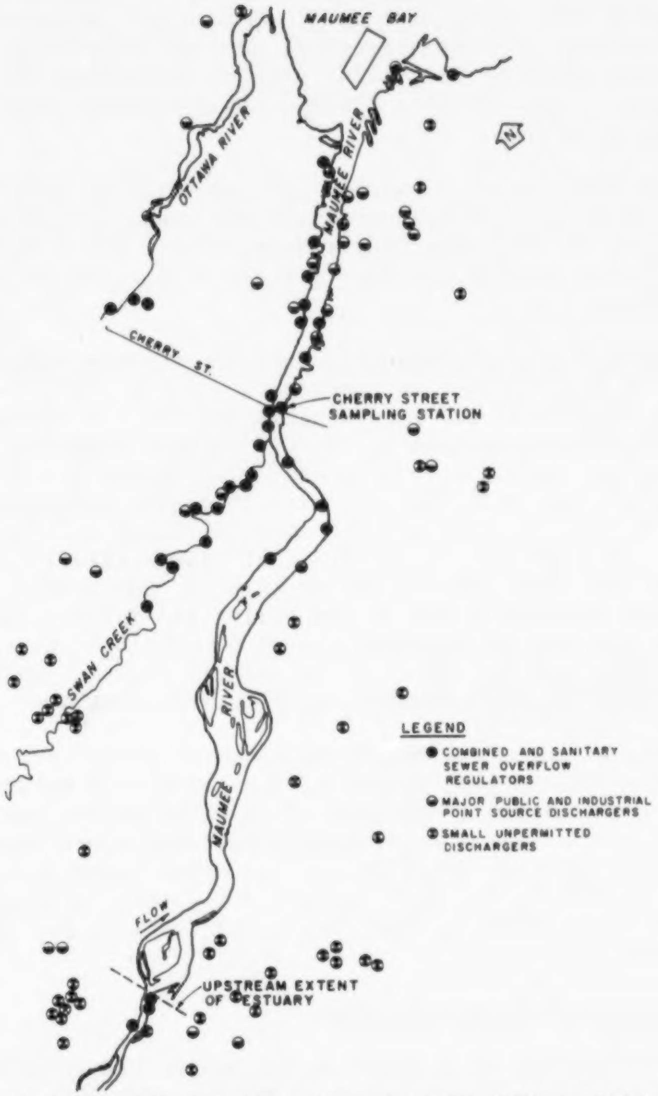
Additional sampling of parameters not measured by LEWMS and TMACOG studies, namely biological oxygen demand and fecal coliform density, has been done by chemists at the Lucas County Maumee River Waste Treatment Plant. Samples which they took in the Maumee River at Waterville have been utilized in this report.

Mass transport studies were conducted in both the Maumee and Cuyahoga estuaries. The Maumee River was sampled at the Cherry Street Bridge in the estuary (river mile 4.8) and above the estuary at the Waterville water treatment plant, 2.0 miles upstream of the USGS water level gage (river mile 21.2). Flow measurements were made at Cherry Street by the U.S. Geological Survey. Samples were collected at three locations on the cross section and two depths. Parameters analyzed are shown in Table 1. The Cuyahoga River was sampled in the estuary at Third Street Bridge (river mile 3.0) and above the estuary at the USGS water level station at Independence (river mile 13.7). Samples were collected on the rising and falling state of the hydrograph by the city of Cleveland Water Quality Laboratory. Estuary flow measurements were made by the U. S. Geological Survey. Methods used for chemical analysis for both studies are reported elsewhere (1).

Figure 2 is a map of the Maumee River study area. The locations of all known wastewater discharges, municipal and industrial permitted and unpermitted, have been marked on the map. Also shown on the map are the locations of 24 of the city of Toledo's combined sewer regulators which discharge to the Maumee River and Swan Creek within the study reach, and numerous "unpermitted" dischargers, which are generally small package wastewater treatment plants discharging less than 50,000 gallons per day of treated sanitary sewage.

Analysis of Estuary Sampling Program

A series of samples were taken on the Maumee River, at Cherry Street Bridge during the storm runoff event of 10 January 1975.


MAUMEE BAY
OTTAWA RIVER
MAUMEE RIVER
N
CHERRY ST.
CHERRY STREET
SAMPLING STATION
SWAN CREEK
MAUMEE RIVER
FLOW
UPSTREAM EXTENT
OF ESTUARY
LEGEND
● COMBINED AND SANITARY SEWER OVERFLOW REGULATORS
● MAJOR PUBLIC AND INDUSTRIAL POINT SOURCE DISCHARGERS
● SMALL UNPERMITTED DISCHARGERS

MAUMEE RIVER ESTUARY WATER QUALITY SURVEY REACH
FIGURE 2

Two samples were taken at each of three locations on the cross section because of the width of the section and the location of bridge piers, as well as the proximity of Swan Creek. After an analysis of velocity profiles indicated that velocities were similar at each location, the samples were composited. The results were three values of flux rate over the storm runoff. These values were then compared with the corresponding hydrograph and mass transport estimate for the Waterville station, about 19 miles up river (Table 2). Interestingly, the mass flux rate is slightly greater in the estuary for corresponding flows indicating no deposition is occurring. The storage retention in the estuary between the two stations shows first a time lag, then a slower recession rate, over the hydrograph (Figure 3).

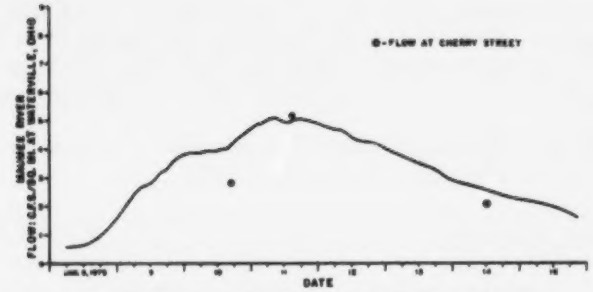
Table 2 - Storm Transport Comparison - Maumee River
(9 thru 14 January 1975)

River Station Waterville Gage (D.A. = 6,329 square miles)					Estuary Station Cherry Street Bridge (D.A. = 6,500 square miles)				
Date	Time	Flow (cfs)	cfs/ sq.mi	Total Phosphorus kg/day	Date	Time	Flow (cfs)	cfs/ sq.mi	Total Phosphorus mt/day
9 Jan 75	1000	17,600	2.78		10 Jan 75	1650	17,800	2.74	28.13
11 Jan 75	1700	32,700	5.16	103.16	11 Jan 75	1430	33,400	5.14	105.97
15 Jan 75	0900	13,000	2.05	19.08	14 Jan 75	1250	13,000	2.00	33,44

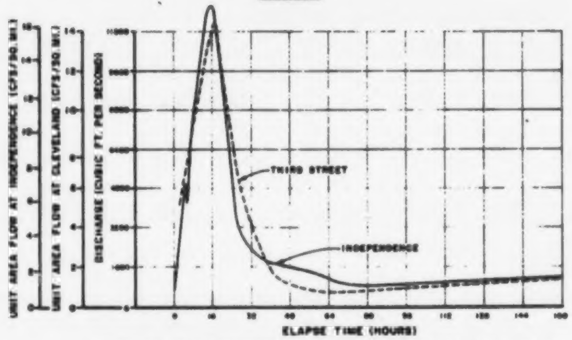
On the Cuyahoga, the estuary was much easier to measure, with the Third Street Bridge station providing a uniform, rectangular channel. The chemical transport however, was far more complicated. During the storm of 21 May through 28 May 1975, simultaneous chemical samples were taken and discharge measurements were made at this station and at Independence, 10.7 miles upriver. Both stations failed to demonstrate a distinct increase in chemical concentration over the hydrograph, with the chemical variations large and erratic for phosphorus as well as for other parameters. Eight discharge measurements and 19 chemical samples were taken at Third Street over a 164 hour period. From these data, eight instantaneous flux estimates were made for total phosphorus. These data were then compared with Independence.

While the concentrations of total phosphorus varied considerably at both stations, the hydrographs of the two stations were nearly simultaneous (Figure 4) and the mass transport past the two stations was of about the same magnitude (Figure 5). Both stations had a maximum discharge of about 12,000 cfs which occurred at 0900 on 22 May. Furthermore, the plots of instantaneous flux versus elapsed time showed both stations carrying a peak flux rate of nearly 12 metric tons per day with approximately equal total mass transport over the runoff period.

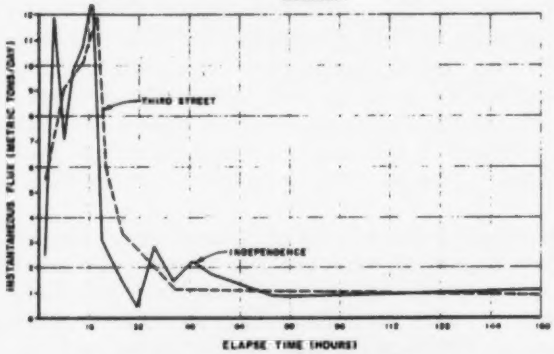
While neither of these analyses can be considered totally conclusive, they do suggest strongly that the measurement of mass transport at riverine



COMPARISON OF FLOW AT WATERVILLE WITH FLOW AT CHERRY STREET
FIGURE 3



HYDROGRAPHS OF CUYAHOGA RIVER AT INDEPENDENCE AND THIRD STREET BRIDGE, CLEVELAND; MAY 21st THROUGH MAY 28th
FIGURE 4



INSTANTANEOUS FLUX RATE OF TOTAL PHOSPHORUS
CUYAHOGA RIVER - MAY 21st THROUGH MAY 28th
FIGURE 5

stations above the estuary sections will give representative estimates of mass flux from the basin during high flow events. The localized effect of urban runoff may be evident much sooner than any rise in stage in the estuary, and so future station sampling should be initiated at the start of the rainfall event. Also, it would appear that no significant storage or discharge of accumulated pollutants occur in the estuary above either station. Since dredging is required in both channels on a large-scale basis, this would appear to be an incorrect conclusion; however, it is likely that the load measured in transport by this sampling is of a different character than that material which fills the channel on the bottom of each river.

Summer Water Quality Surveys

Summer surveys in 1975 were designed to study the water quality of the estuary in response to its complex hydraulics and the inputs of pollution from upstream and local runoff. There were several survey days when stagnation was the apparent cause of extreme water quality degredation in the estuary. 31 July 1975 was the worst, and is here discussed in detail.

Beginning on 19 and 20 July, a heavy rainfall was experienced in the Maumee Basin and locally in Toledo. On the morning of 20 July, 0.97 inches of rain were recorded between 1100 and 1200 at the Toledo Express Airport. A total rainfall of 1.29 inches was recorded between 1000 on the 19th and 1500 on the 20th. This heavy local rainfall flushed a considerable amount of combined sewage and stormwater into the river. At the same time the Maumee River at Waterville was rising from 1,000 cfs on the 18th to 6,600 cfs on the 20th and bringing with it an estimated 324,000 pounds of BOD(5). This fairly large summer storm did not produce enough water to exchange the entire estuary volume and did not significantly alter water velocities in the estuary segment. Velocities here were controlled by the rising and falling of Lake Erie. During this time the Toledo STP continued to discharge at least its long-term average load of BOD(5) of approximately 28,200 pounds per day.

From 20 through 25 July, the Lake Erie stage gage at the mouth of the river was at best jittery, rising and falling only a few tenths of a foot in short period oscillations. Beginning 25 July, wind setups on the lake were somewhat larger causing a 2.0-foot rise on the evening of 25 July. Immediately the lake level fell and soon began the very regular oscillations which characterize the Lake Erie seiche. This seiche was not sustained, but damped until the day of sampling when the stage was virtually flat. By this time the discharge of the Maumee River at Waterville had fallen to 500 cfs, which was nearly as low as it would fall all summer.

The combination of factors during the previous 10 days was adding up to a very bad time for the river. It had received a heavy load of oxygen demanding substances from both the local urban area and the rest of the Maume Basin from storm runoff. River discharge was now low and the lake was quiescent. It was very hot and river temperaures were as warm as they would be all summer.

Figure 6 depicts dissolved oxygen, temperature, and conductivity data gathered on 31 July 1976. All parameters measured showed marked stratification. Dissolved oxygen is stratified by 6.2 mg/l, 7.2 mg/l at the surface, 1.0 mg/l near the bottom. There was no discernable sag point. Minimum DO values were observed at RM 2.5 and 8.5. Temperature was vertically stratified by 2 to 3°C at most points in the river.

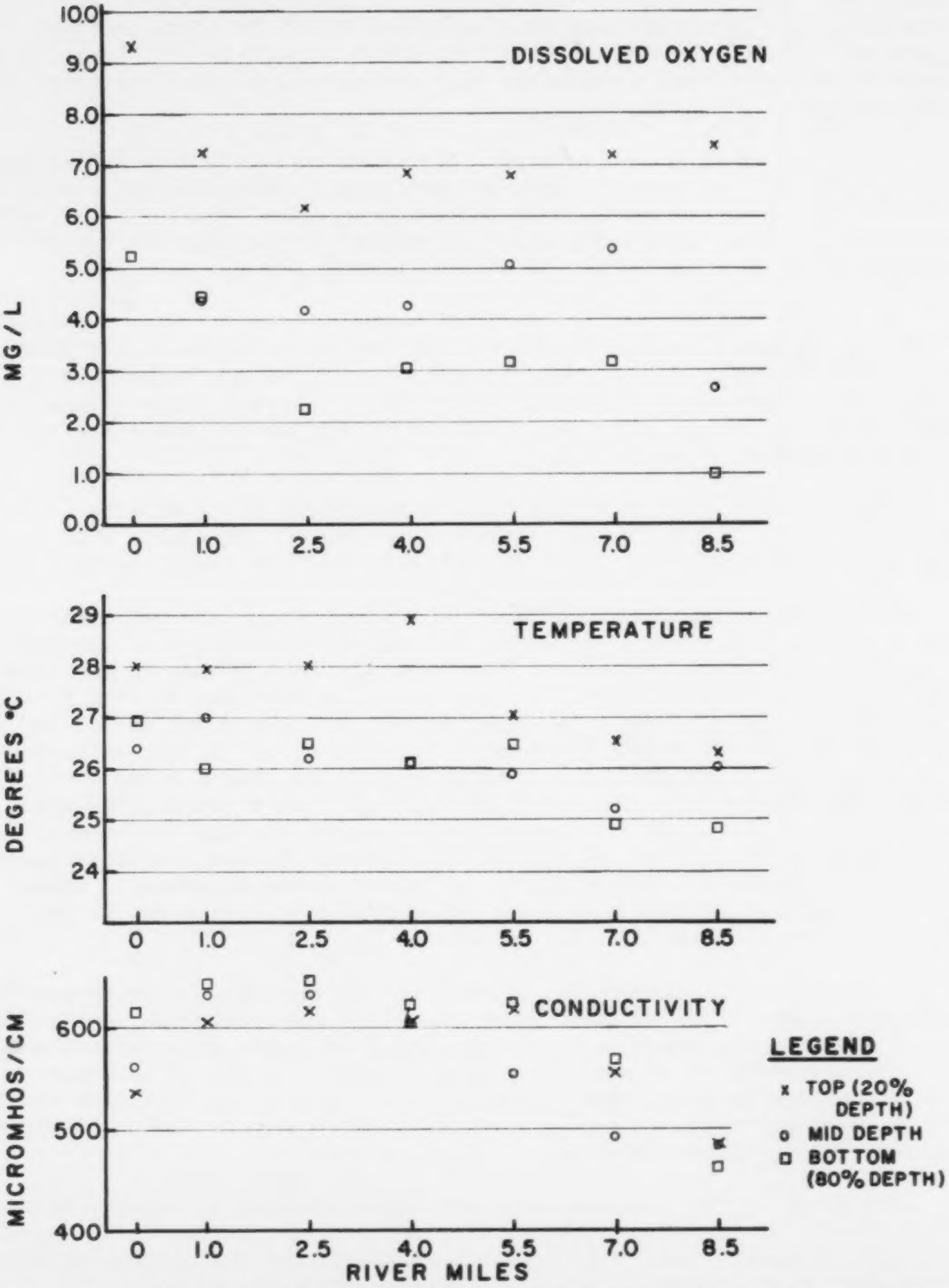
Conductivity was higher throughout the river than on any other sampling day of the summer, and was highest near the mouth of the river above the Toledo STP outfalls. Maximum conductivity readings, 630 umhos/cm, are apparent at RM 1.0 on the west bank of the river, just upstream of the Toledo STP outfalls, and dropping off in all directions from that point. At RM 0.0, conductivity is reduced to 560 umhos/cm as boundary waters of the Maumee River and Bay mix. At RM 8.5, conductivity has fallen to 470 umhos/cm as pure river water enters the survey reach. Figure 11 shows that conductivity of the Maumee at Waterville had been as low as 380 umhos/cm on 18-20 July. It is apparent that the high conductivity effluents of the Toledo STP and secondarily the Gulf Oil Refinery are influencing the dissolved solids content of the entire estuarine volume below RM 8.5.

Such severe stratification occurred only on sampling dates that were preceded by a quiescent lake. Stratification, to some degree, was almost always apparent, but all other sampling dates had been preceeded by at least the regular mixing action of the seiche.

The Upstream Heritage and its Influence on Estuarine Water Quality

This section discusses water quality measurements upstream at the Waterville gage and then relates it to water quality measured in the estuary. This relationship varies at different times of the year, with different concomitant hydrologic regimes, and in different hydraulic segments. The flows considered are high flows associated with storm events, intermediate flows and low flows associated with the dry months of the year (but not necessarily drought flows).

The bulk of the pollutants coming out of the Maumee Basin is transported during a few large storm events which take place primarily during the winter and early spring of the year. Approximately 80 percent the total annual transport of sediment and phosphorus take place during 20 percent of the time. The bulk of nutrients and oxygen demanding substances are also transported at this time. During these high flow winter storms, water temperature is low and reaeration is high so the pollutants do not have a strong impact on the traditional parameters of water quality, but they do have their impacts at other times and in distant locations. As shown previously material is transported through the estuary during high flow events. However, during runoff events associated with lower flows some of the nutrient and oxygen demanding sediment settle as the storm waters of the



DISSOLVED OXYGEN, TEMPERATURE & CONDUCTIVITY FOR MAUMEE RIVER SURVEY REACH 31 JULY 1975

FIGURE 6

Maumee flow into the estuary where the river deepens and widens and velocities decrease. Although this is a small amount of the total sediment transported by the Maumee, it does affect water quality. As the water warms later in the year, these sediments will demend oxygen and cause water quality degradation.

There are other direct impacts. During storm events, the Maumee may carry 2,000 mg/l or more of suspended sediments. These sediments must be removed for the water supplies of Defiance, Napolean, Bowling Green, and Waterville. Also, nitrate exceeded the USEPA recommended maximum concentration of 10.0 mg/l NO_3-N on 14 days during calendar year 1975.

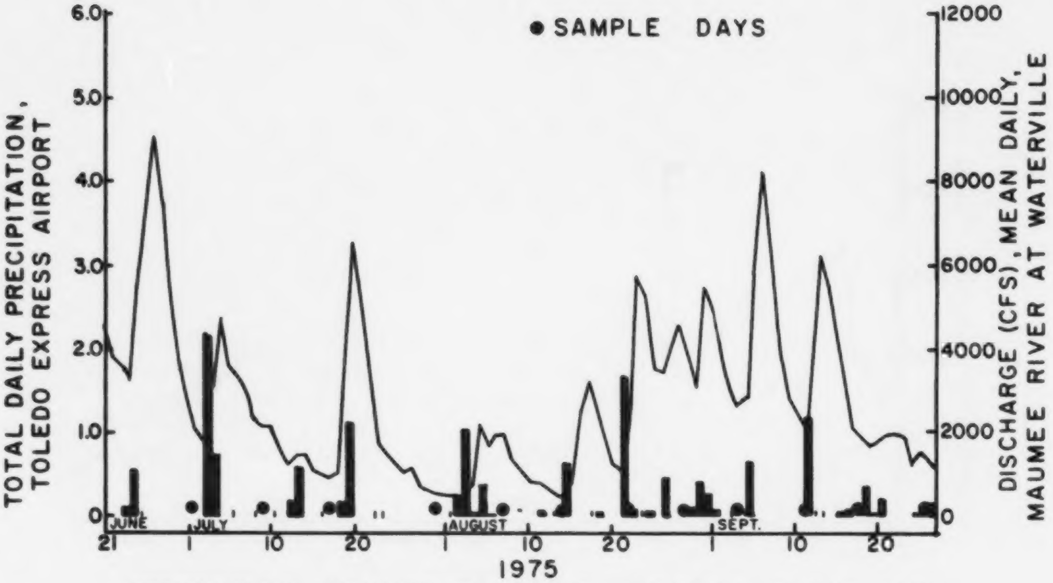
While the loadings are primarily of importance to the long-term quality of the Great Lakes system, concentrations and their instantaneous time of occurrence are most significant for water quality in the riverine and estuarine segments. As discussed in the section on estuarine stagnation, lake level variations and river discharge velocities are important variables to instantaneous water quality.

Figure 7 is a plot of stream flow of the Maumee River at Waterville and rainfall at the Toledo Express Airport near Toledo. The dates when sampling was done in the estuarine reach are given by the dots along the date scale.

During the summer months, flows are reduced and so are pollutant concentrations. The river could always be considered enriched or eutrophic with respect to the concentrations of nutrients for which standards have been set for Lake Erie. Figures 8 thru 12 plot the concentrations of five of the parameters which were measured daily (except for some downtime) in the Maumee River at Waterville during the survey period of 1975. Nitrite-nitrate nitrogen (Figure 8) shows more of a seasonal dependence than a storm event dependence. The abrupt increases correlate well with storm events, but the greatest variance is with a longer time extending through the summer. It is most interesting to note that large increases in nitrate concentration do not take place during the storms which took place during September. Ground cover is most complete at this time and most available soil nitrogen is probably incorporated into growing plant material.

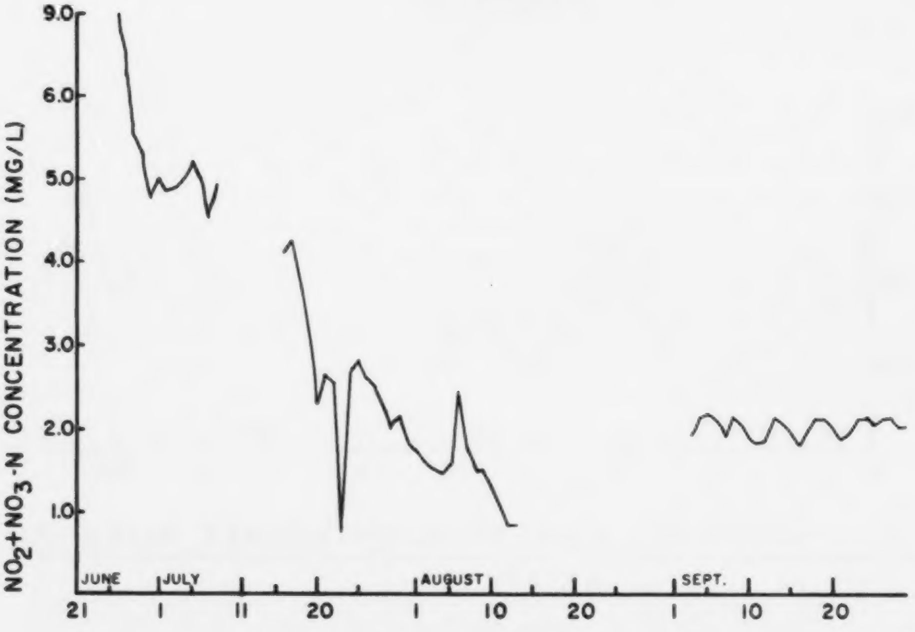
The suspended sediment plot shows a similar pattern, but with stronger flow dependence. The sediment graph parallels the hydrograph very closely, but a seasonal variation is apparent. The late summer storm hydrographs do not produce sediment graphs of the same magnitude as the early summer storms. The difference is small, but obvious. Again, ground cover is more complete. It would appear that although ground cover does reduce sediment yield, the protection is not complete.

The first of the high velocity late summer storms, 23 August, is smaller than several of the later events, but produces the greatest peak suspended sediment concentration. This concentration reflects the washout of sediments from short-term storage in the river bottom during the summer. The 6 September event produces only a small increase over the elevated background concentration and is perhaps a better indication of sediment runoff which can



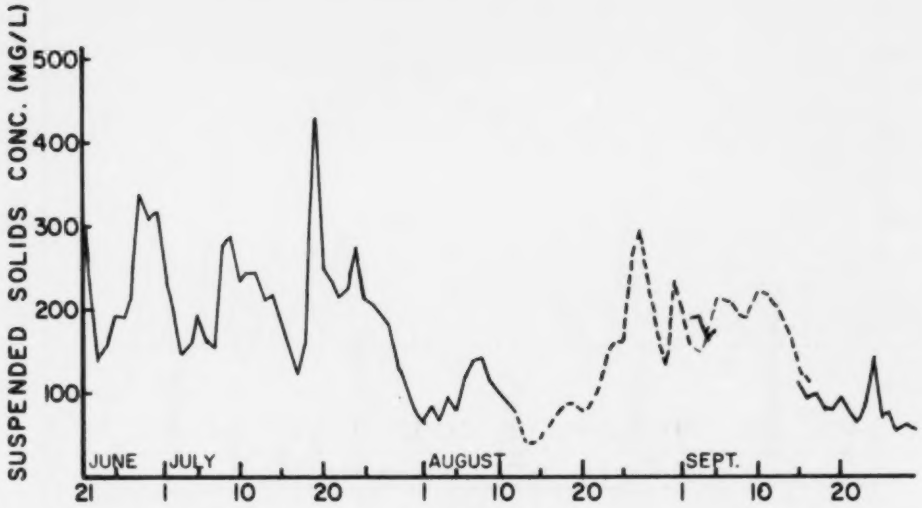
STREAMFLOW, MAUMEE RIVER AT WATERVILLE AND TOTAL DAILY PRECIPITATION, TOLEDO EXPRESS AIRPORT

FIGURE 7



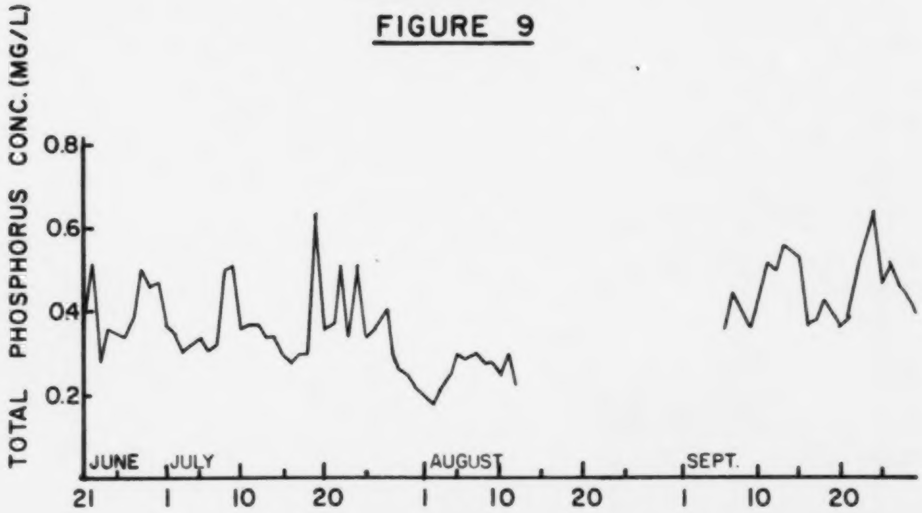
NITRATE + NITRITE-N, MAUMEE RIVER AT WATERVILLE

FIGURE 8



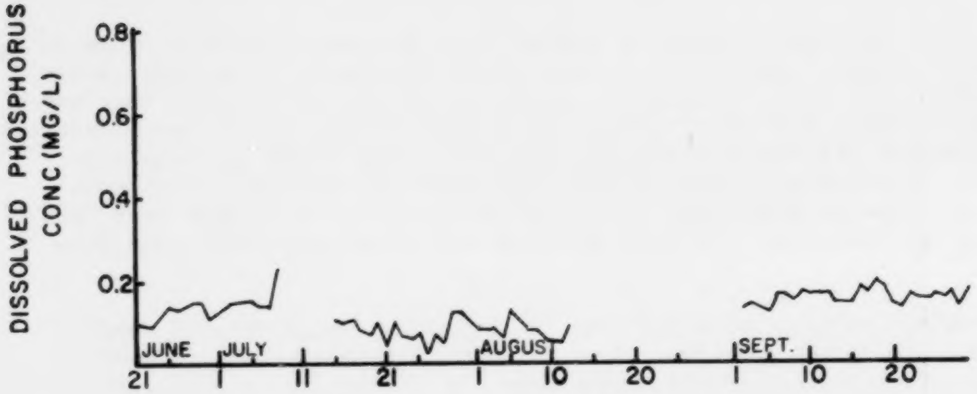
SUSPENDED SOLIDS CONCENTRATION, MAUMEE RIVER AT WATERVILLE

FIGURE 9



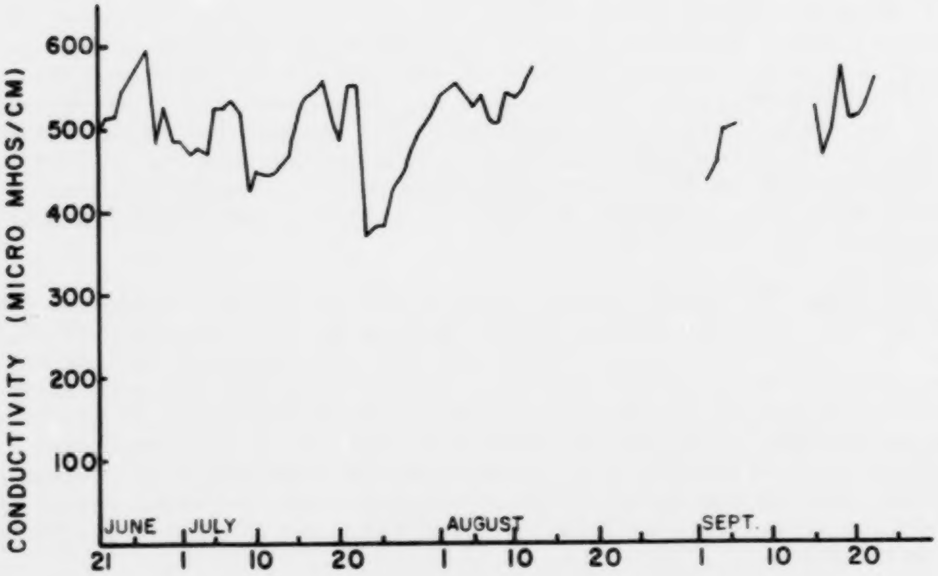
TOTAL PHOSPHORUS CONCENTRATION, MAUMEE RIVER AT WATERVILLE

FIGURE 10



DISSOLVED PHOSPHORUS CONCENTRATION, MAUMEE RIVER AT WATERVILLE

FIGURE 11



CONDUCTIVITY, MAUMEE RIVER AT WATERVILLE

FIGURE 12

be expected from a late summer storm. It is also, perhaps, a better indication of the effectiveness of ground cover protection in late summer.

The plot of total phosphorus demonstrates the same flow dependence as suspended sediments and indicates that total phosphorus is strongly associated with land wash. Dissolved phosphorus, on the other hand, does not increase markedly with flow during summer storm events and in many instances shows a slight dilutional tendency. Concentrations never decrease sufficiently to decrease loads. Rather, the loads of dissolved phosphorus always do increase with flow. This may be associated with land wash (both urban and agricultural) and with combined and storm sewer overflows from cities.

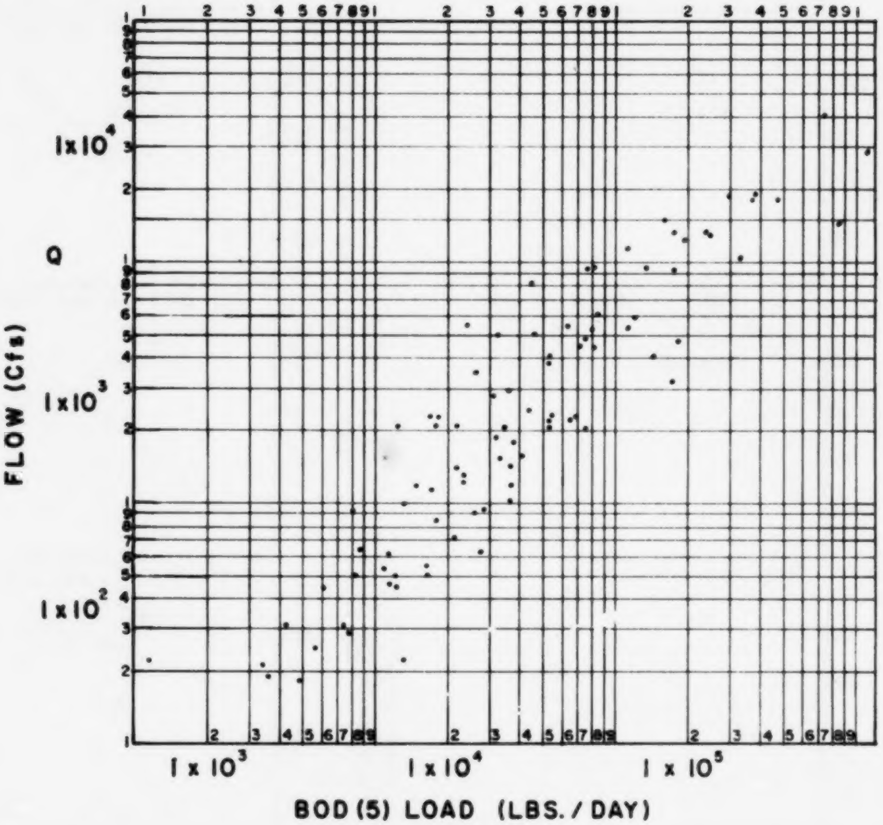
Dissolved phosphorus is the form of phosphorus which is most readily available as a plant nutrient. Over the entire summer period, dissolved phosphorus was present in the Maumee River at Waterville at a concentration of 0.136 ± 0.009 mg/l (mean and standard error for 105 samples). This concentration indicates that there was almost always plenty of phosphorus in the river for plant growth. The major source of this important plant nutrient during the summer months was in the continuous input associated with municipal sewage treatment plants and equilibrium dissolution from bottom sediments.

Conductivity, as shown in Figure 12, varied inversely with flow and did not exhibit a seasonal dependence.

Figure 13 is a load graph of flux versus flow for BOD(5) (20°C, unseeded, dark) in the Maumee River at Waterville over the period from September 1974 through April 1976. The data used for this graph was gathered as part of an NPDES self monitoring program by staff of the Lucas County/Maumee River Wastewater Management Plant. The samples are taken once weekly at Waterville very near the location used for sampling in this study. The shape of the curve indicates a very strong relationship between flow and the load of BOD(5) being carried in the river. When flow goes up, the BOD(5) load increases. When water runs off the land it carries oxygen demanding substances with it.

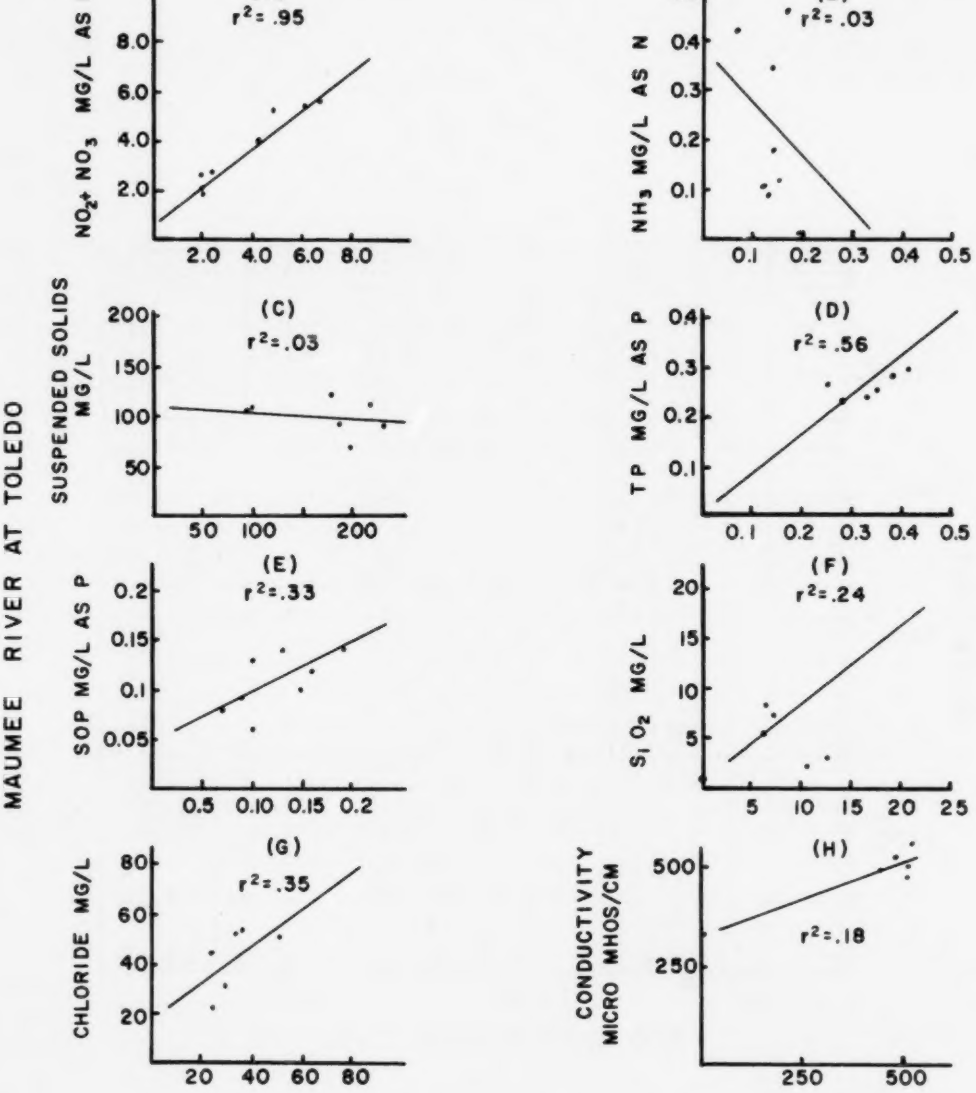
At mean flow, the Maumee carries from 25,000 to 200,000 lbs/day of BOD(5) compared to the long-term average BOD(5) loading of the Toledo STP of 28,700 lbs/day. At higher flows above 10,000 cfs, the data does not include any load measurements below 100,000 lbs/day. Considering that the population of the remainder of the entire Maume River Basin above Waterville is little more than the population of the Toledo urban area, that it is scattered over 6,330 square miles and that BOD(5) is a nonconservative substance which decays with time and distance in the river, it seems unlikely that the major source of this pollutant is people. Agricultural land wash must be considered to be a significant source.

Figure 14 shows the relationship of water quality in the Maumee River at Waterville to the quality of the waters of the lacustrine estuary of the Maumee at Toledo. The analysis was performed with water quality at Waterville as the independent variable, and water quality in the estuary as



LOAD GRAPH OF BOD (5) vs. FLOW, MAUMEE RIVER AT WATERVILLE

FIGURE 13



RELATIONSHIP BETWEEN THE WATER QUALITY OF THE MAUMEE RIVER AT WATERVILLE (ABSCISSA) AND THE MAUMEE ESTUARY AT TOLEDO (ORDINATE) JULY TO SEPTEMBER 1977;

(A) NITRITE - NITRATE NITROGEN	(B) AMMONIA NITROGEN
(C) SUSPENDED SEDIMENT	(D) TOTAL PHOSPHORUS
(E) SOLUBLE ORTHO PHOSPHORUS	(F) DISSOLVED SILICA
(G) CHLORIDE	(H) CONDUCTIVITY

ABSCISSA AND ORDINATE ARE THE SAME SCALE.

FIGURE 14

the dependent variable. The concentrations representing water quality in the estuary are an average of measurements made in the estuary for one day. Six stations with samples taken at the middle of the river at mid-depth were used. The stations were averaged to integrate all sources of input to the estuary and the effects of time of passage through the estuary. The concentrations representing water quality at Waterville are the mean of each parameter for 10 days prior to the day of sampling in the estuary.

The 10 day mean was used to allow for time of travel and mixing between Waterville and Toledo. This travel time is not an unrealistically long time due to the relatively low flow volumes and the large estuarine volume.

From Figure 14a it can be seen that nitrite-nitrate nitrogen concentrations in the estuary are closely related to concentrations at Waterville. This close correlation ($r^2 = .95$) indicates the major contributor of nitrite-nitrate nitrogen to the estuary is from the upstream heritage. Ammonia concentration at Toledo are independent of Waterville and are usually higher (Figure 14b). This increase of concentrations indicates the existence of local inputs of ammonia to the estuary. Suspended solids (Figure 14c) at Toledo are independent of and usually less than those at Waterville. This decrease indicates settling of solids during the low flow summer period. However, since suspended solids in the estuary reaches a minimum of about 100 mg/l independent of the Waterville concentration, it seems for this period that the fine clays which remained in suspension remained relatively constant at the flows observed.

Total phosphorus concentrations on Figure 14d are well correlated ($r^2 = .56$). However, some settling of phosphorus attached to suspended sediment did occur. Soluble orthophosphorus concentrations (Figure 14e) are greater at Waterville than in the estuary. This loss seems to indicate some biological uptake occurring in the estuary. Soluble silica concentrations (Figure 14f) are poorly correlated between Waterville and the estuary. Chloride concentrations (Figure 14g) are greater in the estuary than at Waterville and indicate sources of local input to the estuary. Conductivity measurement (Figure 14h) do not show much variation between the estuary and Waterville.

CONCLUSION

This analysis demonstrates that the estuarine reach of the Maumee River is impacted by pollutants generated in the remainder of the basin; that the water quality problems in the Toledo urban area are not solely the results of Toledo's own pollution of the river. Inputs of pollutants to the river during even moderately small summer storms can be considerably larger than Toledo's inputs. Impacts are increased by the fact that the entire volume of such summer storms does little more than replace a single volume of the estuarine volume. This polluted water often becomes stagnant in the estuary for many days sloshing back and forth over the organically enriched bottom sediments. In warm weather these events multiply to severely degrade the quality of water in the Maumee River.

During low flow periods and small events the estuaries act as traps for suspended solids and total phosphorus. However, the bulk of this material is transported during a few high flow events when it is carried directly through the estuaries.

REFERENCES

1. U. S. Army Corps of Engineers, "Lake Erie Wastewater Management Study, Preliminary Feasibility Report," 3 Vol, Buffalo, NY, December 1975.

WATER QUALITY AND URBAN RUNOFF IN SELECTED CANAL COMMUNITIES ALONG THE TEXAS COAST

by

Allen L. Messenger
Hays & Lindsey, Inc.
Austin, Texas

and

Tom D. Reynolds
Texas A&M University
College Station, Texas

ABSTRACT

Water and runoff samples from seven waterfront communities in the Galveston Bay area were collected and analyzed in order to evaluate causes of canal water quality problems. Until present, community design has been based on optimum utilization of land area with little or no regard toward the effect of development on water quality. The primary cause of water quality problems in these communities appears to be urban runoff.

Samples of canal waters were collected over a five-month period and analyzed for nutrients, oxygen damand, pesticides, and hydrological variables, including Rhodamine dye concentrations. In general, canal waters exerted BOD_5 values of 2-10 mg/l with no problems associated with toxic substances. Evaluation of domestic wastewater data from centralized treatment facilities indicates that these wastewater streams are not major sources of pollution loading.

Runoff samples were collected from three rainfall-runoff events and were found to contain significant amounts of carbonaceous material. Also, the possibility of canal sediment resuspension by point discharge of runoff was investigated using a canal model. The two-year frequency rainfall event for the Galveston area was found to produce significant resuspension of high BOD_5 benthic sediments. These data were applied using the modified Streeter-Phelps equations for estuarine dissolved oxygen analysis and were found to fit actual conditions with reasonable accuracy.

INTRODUCTION

During recent years population increases along the Texas coast have resulted in a corresponding increase in housing demand. Most housing requirements are being met by conventional urban developments. However, the proximity of bay waters along the Texas coast allows urban planners the unique opportunity to design and build waterfront communities. The basic purpose in the design of these subdivisions is to provide each lot with one side of waterfront footage. Access to the surrounding waters is through a canal network which opens into the parent body, typically a bayou, bay, or saltwater lake.

At last count, there were almost 60 coastal waterfront communities along the Texas coast. They represent a wide range in size, design, and cost. For example, Bayou Vista located in La Marque has 1318 lots, approximately 40,000 feet of canals of dead-end design, and homes in the $60,000 range. In contrast, Jarbo Bayou has 80 lots, approximately 5,000 feet of canal with continuous flow provided by Jarbo Bayou, and the homes sell for over $100,000.

There are problems associated with canal communities which, although independent of price, are dependent on the design and size of the community. In several of the larger communities, utilizing canal networks of dead-end design, fish-kills and dangerously high coliform and enteric virus concentrations have occurred (1). As a result of the latter condition, the public health department has banned the harvesting of oysters from the canals. In addition, the safety of swimming in the canals has been questioned.

Seven communities in the Houston-Galveston area were chosen for study in order to investigate the interrelationship between pollutional loading and canal water quality. One basis on which these communities are differentiated is domestic wastewater handling and treatment. One of the communities utilizes septic tank systems while the others have wastewater collection systems and centralized domestic wastewater treatment. Urban runoff, another major factor influencing canal water quality, is handled in one of two ways. In some designs, runoff is allowed to run overland into the canal along the entire canal length. In other designs, runoff is collected in open ditches or storm sewers and then discharged at points along the canal.

LITERATURE SURVEY

The most comprehensive report found on the natural waters surrounding most of the study areas is "The Galveston Bay Study" by Gloyna and Malina (2). Included in the report is information on all the bays in the Galveston Bay system. West Bay is the outer bay in the Galveston Bay system, and as a result, has been impacted the least by the rapidly growing Houston-Galveston area. Six of the communities studied are on West Bay. According to this report, the waters of West Bay are relatively unpolluted. It is reported that samples taken from the bayous and along the sides of the bay indicate pollution loads from domestic sources were common enough to cause the water to exert a BOD_5 of 2 to 4 mg/l.

Trent and Lindall (3) present a summary of recommendations concerning coastal canal development including:

I. Minimize disruption of the existing habitat as much as possible.
   1) Restrict community development to non-wetland areas.
   2) Access canals to communities' canal systems should be by the shortest and least environmentally damaging route possible.
   3) Turbidity and sediment dispersion should be controlled as much as possible.
      a) Complete all construction prior to connecting the system to the surrounding natural waters.
      b) Access canal construction should be conducted in a manner which minimizes sediment dispersion.
      c) Spoil disposal sites and easements should be reserved for future canal maintenance.

II. The canal systems should be designed to maintain state and federal water quality standards, especially for dissolved oxygen.

Another paper covering the effects of coastal development on water quality, and plant and animal productivity has been published by Taylor, et al. found that canal bottoms are soft and may exceed 90-percent silt and clay whereas in undredged areas, the bottom is up to 94-percent sand and shell. They found the silt and clay bottoms in the canals were "unsuitable for most bottom invertebrates found in other parts of the bay." They also state of a 10-year old canal system: "In 10 years, recolonization of the canal sediments has been negligible and it appears doubtful that soft sediments of bayfill canals will ever support a rich or diverse infauna "(4).

Moore and Trent (5) monitored one of the canal systems studied for this report. They were concerned mainly with the canal's effect on the oyster, Crassostrea virginica. The study is unique because it was aimed at utilizing canal communities for oyster production. However, low dissolved oxygen levels in the canals resulted in a high mortality among the oysters making it unsuitable for oyster production.

In a report by Paulson (6), some of the most comprehensive work to date is presented on basic water quality in canal communities. Coliforms were found to increase while dissolved oxygen levels decreased with increasing distance away from the canals' inlet. The BOD_5 levels were low and no differences were found between natural and altered areas. Flushing rates were found to be the same in both manmade and natural systems. It should be noted that in Paulson's work, the canals under study were shallower than the adjacent water bodies. This condition is the opposite of the conditions existing in the Galveston area.

RESULTS AND DISCUSSION

The communities selected for study incorporate wide variations in design. Design variations include: canal length, inlet design, domestic wastewater handling and treatment, and runoff routing. Data collected from seven canal communities are presented. Comparisons are made based on general differences in deisgn and two communities are used to examine the effect of canal age and wastewater handling on water quality. Hydrological data include TOC, BOD_5, salinity, conductivity, temperature, coliforms, Kjeldahl nitrogen, nitrite, nitrate, phosphates, dissolved oxygen, and Rhodamine dye concentrations.

A total of seven communities were monitored and the resulting data are presented in Table I. However, more detailed data will be presented on only two communities. For further information and data analysis refer to the thesis from which this paper was generated (7).

Effect of Canal Age

Figures 1 and 2 are graphic comparisons of hydrological data collected in the three sections of various ages in Bayou Vista (Figure 3). These figures compare various hydrological variables and indicate only slight differences in concentrations with no trends apparent. These data compare well with previous work by Hall (8).

It should be noted from Figure 3 that the oldest section is located where it has the best flushing and that the newer canals are located where they have minimal flushing. Nutrient levels in the 24-year old canals averaged lower (refer to Figure 2) than the newer sections. Similarly, TOC and BOD_5 concentrations varied only slightly between sections (Figure 1) indi-

Table I Summary of Water Quality and Runoff Data

Community	Bayou Vista		Sea Isle		Jamaica Beach		Pirates Cove		Jarbo Bayou		Tiki Island		Spanish Grant	
Parameter	C*	R**	C	R	C	R	C	R	C	R	C	R	C	R
BOD_5, mg/l	4.8	8.4	3.4	11.5	4.0	10.5	3.8	9.9	4.8	-	3.5	13.4	3.0	-
TOC, mg/l	18.1	25.4	20.7	22.3	21.4	30.4	24.7	15.1	23.2	-	26.9	19.5	13.8	-
Conductivity, umhos	20.9×10^4	-	25.6×10^4	-	27.2×10^4	-	29.1×10^4	-	8.7×10^4	-	22.3×10^4	-	25.3×10^4	-
Salinity, %	15.0	-	19.6	-	20.4	-	19.9	-	5.9	-	18.3	-	18.1	
Coliforms, MPN	1009	-	77.0	-	85.0	-	62.0	-	2859	-	47.0	-	8.0	-
Fecals, MPN	482	-	12.6	-	28.0	-	63.5	-	1126	-	11.0	-	6.0	-
TKN, mg/l	1.02	1.45	0.88	1.19	1.08	1.28	1.14	2.04	1.04	-	1.00	1.36	0.9	-
NO_2-N, mg/l	0.06	0.20	0.04	0.11	0.05	0.09	0.05	0.42	0.06	-	0.02	0.24	0.02	-
NO_3-N, mg/l	0.54	0.28	0.24	0.63	0.20	0.16	0.17	0.38	0.80	-	0.34	2.20	0.10	-
Phosphates, mg/l	0.48	0.24	0.22	0.28	0.17	0.19	0.17	0.51	0.52	-	0.30	0.36	0.10	-

*C - Canal Water **R - Runoff

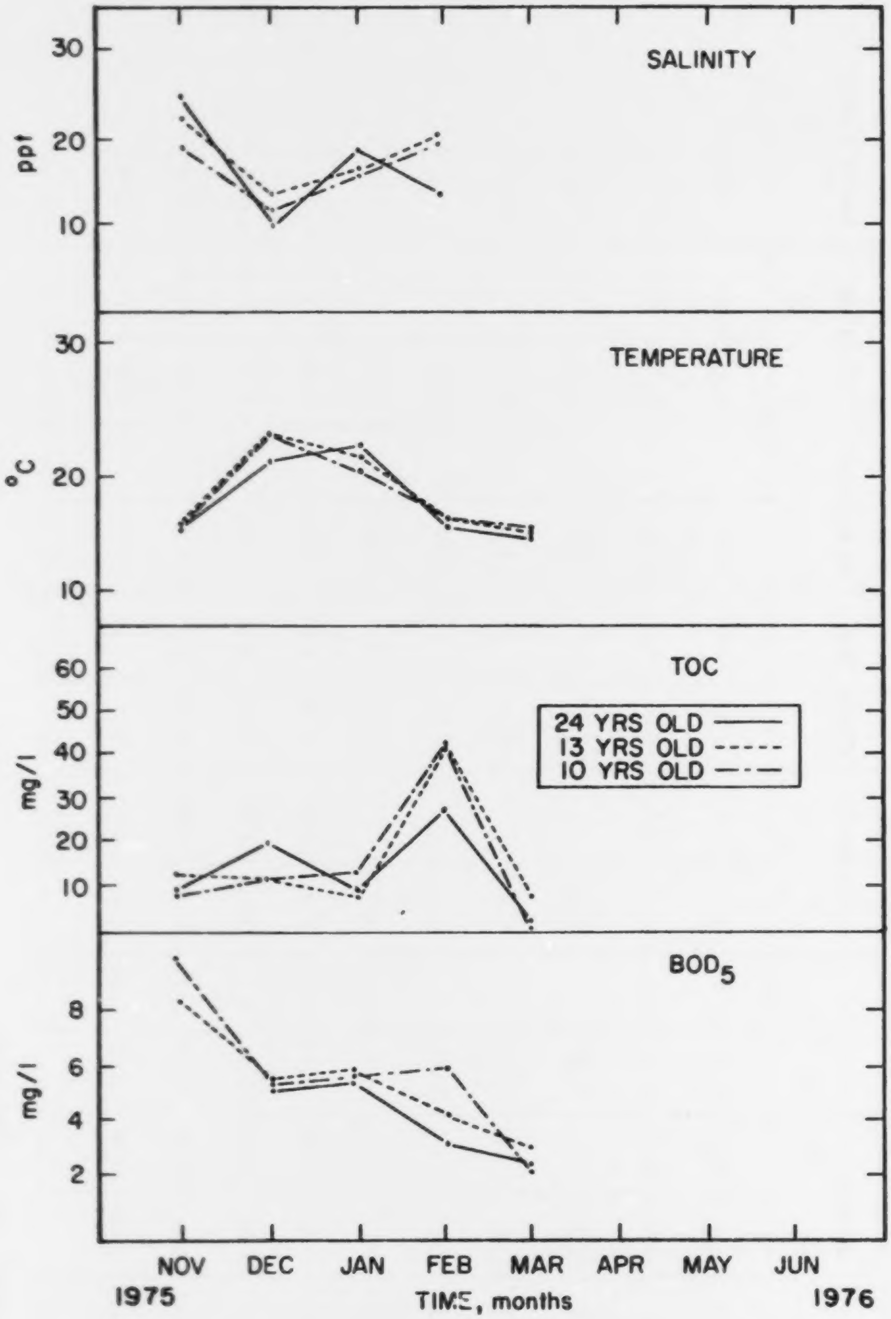


FIGURE 1. — COMPARISONS OF HYDROGRAPHIC VARIABLES IN CANALS OF VARIOUS AGES IN BAYOU VISTA

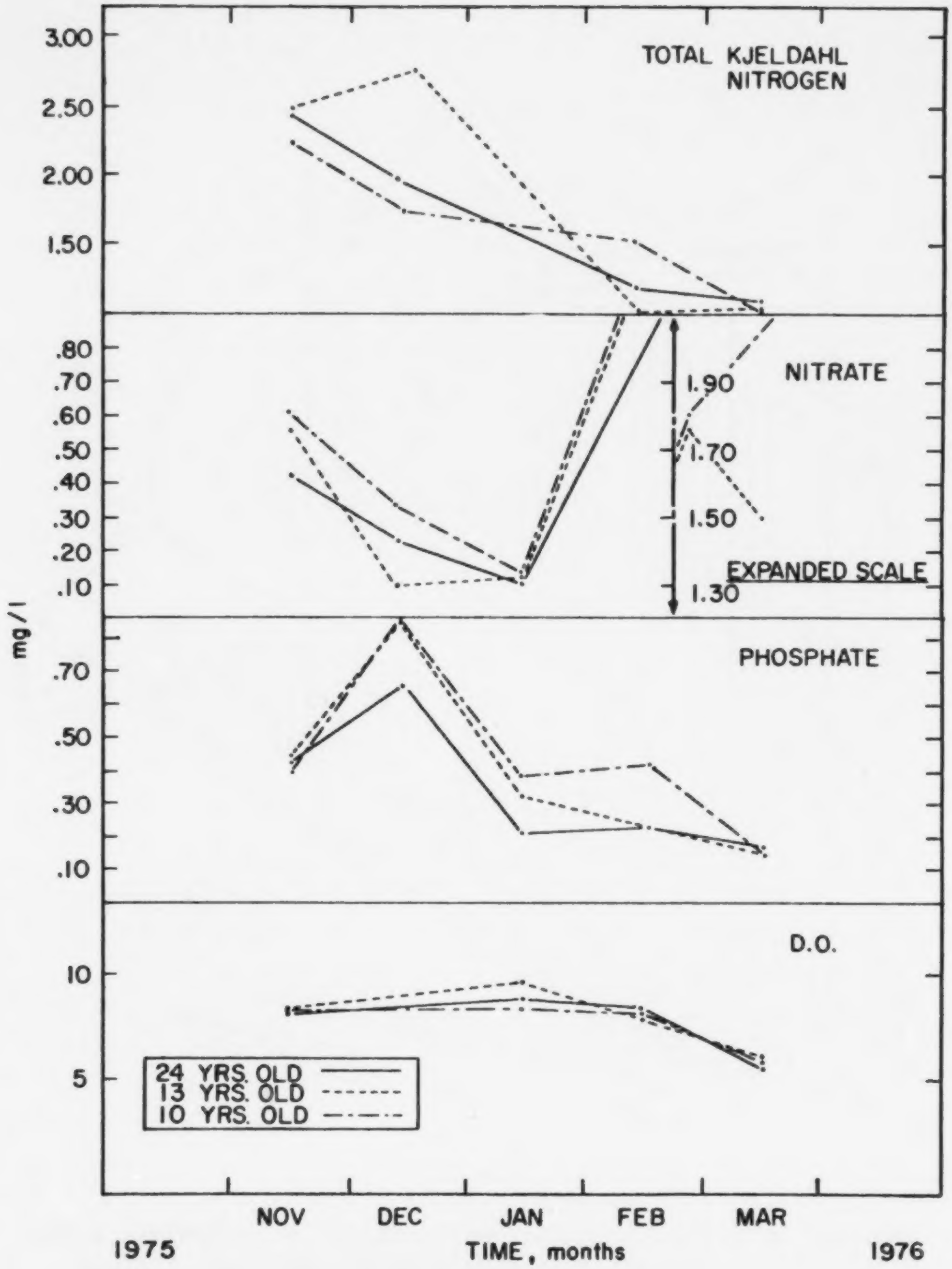


FIGURE 2. —COMPARISON OF NUTRIENTS AND D.O. IN CANALS OF VARIOUS AGES IN BAYOU VISTA

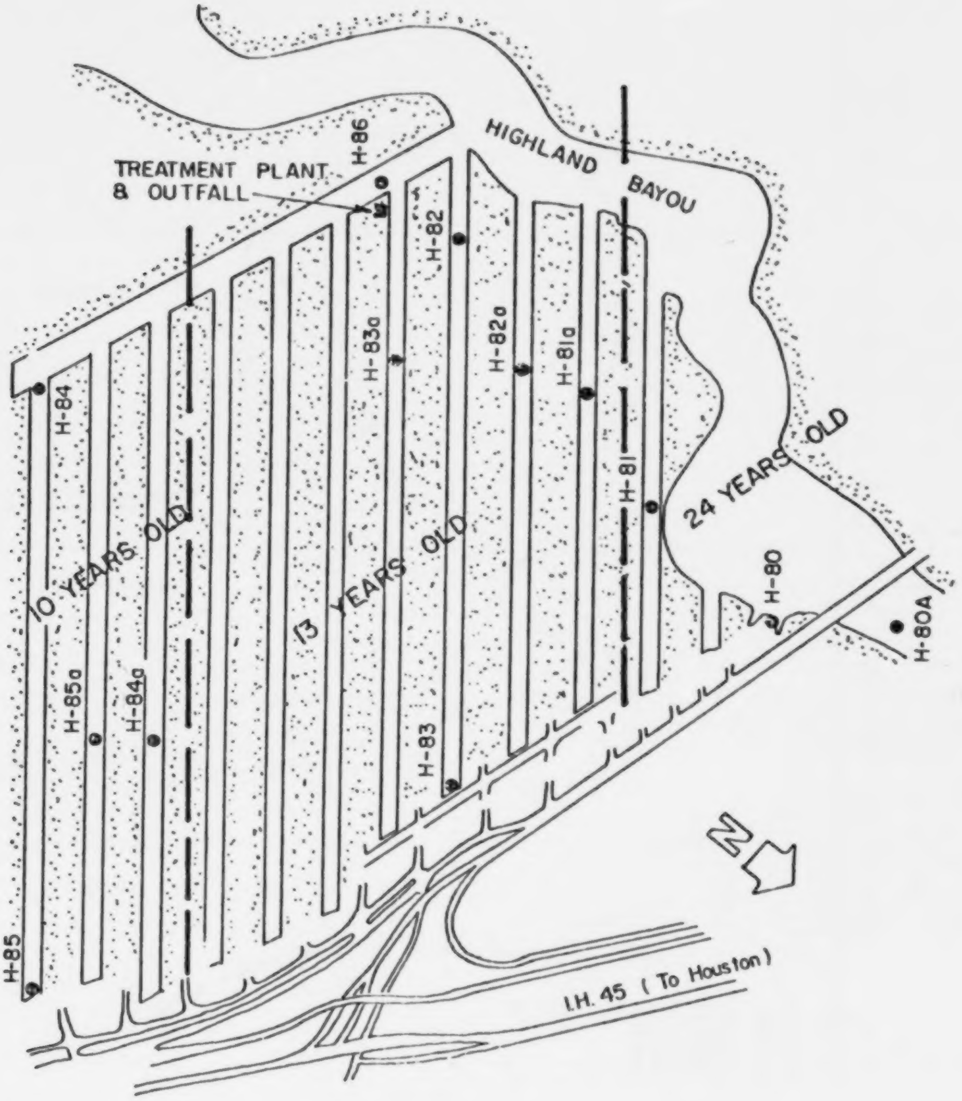


FIGURE 3. - BAYOU VISTA CANAL SYSTEM SHOWING THREE STAGES OF DEVELOPMENT AND THE STATION LOCATIONS.

cating that canal age is not a dominant factor in determining canal water quality when flushing and circulation are also exerting an effect.

An additional variable differentiating between the three sections is canal depth. Section I (10-years old) ranges 9.5 to 12 feet (2.9 to 3.7 m) in depth. Section II (13-years old) ranges from 5 to 11 feet (1.5 to 3.4 m) deep with the 5 foot (1.5 m) depth occurring at the wastewater treatment plant outfall. Typically, the minimum depth in Section II is 8 feet (2.4 m). Section III (24-years old) consists of two smaller canals of about 8 to 10 feet (2.4 to 3.0 m) in depth. All depths are from mean sea level and are approximations.

Bayou Vista

Bayou Vista, located in La Marque, contains 1318 lots and 540 homes. The canals are of dead-end design and are approximately 3400 feet (1036 m) in length. The lots have 50 feet (15.2 m) of waterfront footage and are 90-feet (27.4 m) deep. All homes are elevated on pilings.

Bayou Vista consists of three sections of varying age as shown in Figure 3. The section with sample stations H-84 and H-85 is 10-years old. The section with stations H-82 and H-83 is 13-years old and the section with station H-81 is 23-years old. Station H-80 is located on Highland Bayou outside the community.

The domestic sewage is collected and treated by a wastewater treatment facility and is discharged at station H-86. Discharges averaged 97,000 gallons/day (367 m^3/day) over February and March 1976 with a BOD_5 of 27 mg/l and suspended solids of 85 mg/l.

Storm runoff is removed by two methods. Runoff from streets and front yards is collected in ditches and discharged at points along the canals. Runoff from the back yards (canal footage) is allowed to flow over the bulkheads.

The canal on which station H-83 is located has a history of Gulf Menhaden (*Bravoortia patronus*) fish kills. The kills occur after rainfall during the critical summer months and, according to canal residents interviewed, at night or early morning. This indicated a kill caused by dissolved oxygen depletion in the canal water due to an algal bloom and/or the resuspension of canal sediments.

It should be noted that one of the most popular forms of recreation in Bayou Vista is back-yard fishing, both at night and during the day. At night, the fishermen shine powerful batteries of spotlights into the canal waters and fish with good success. Catches of 15 to 20 game fish have been recorded photographically. Late night monitoring revealed oxygen levels of 12.5 mg/l under these lights although surrounding waters were generally less than 4.0 mg/l. Attraction of fishes is by photo- or oxygen tropism.

Canal waters averaged 5.3 mg/l BOD_5, 10.1 mg/l TOC, 1.98 mg/l total nitrogen, and 0.586 mg/l orthophosphates. Nutrient levels in the runoff samples averaged slightly lower than the canal waters and BOD_5 levels averaged 58-percent higher in the runoff samples.

Dissolved Oxygen Profile

The largest single problem with the canals in any canal community is oxygen depletion in the canal water. There are several contributing factors:

1) low or non-existent circulation,
2) poor tidal flushing,
3) loading into the canals from runoff, and
4) loading into the canals from septic tank and treatment plant wastewaters.

The oxygen depletion which results from certain combinations of these factors is the result of an increased algal population which produces appreciable amounts of oxygen during the day but likewise consumes large amounts of oxygen at night.

Figure 4 is a graphic summary of dissolved oxygen data collected in Bayou Vista during July. It was collected from 11:00 p.m. to 1:00 a.m. The top graph shows the dissolved oxygen concentration at surface and the lower graph indicates the depth at which the oxygen level was less than 1.0 mg/l.

The data clearly show the relationship between canal length and dissolved oxygen concentration. The further down the canal from the inlet, the lower is the concentration. In all three canals there was greater than 1 mg/l of oxygen at the inlet at all depths monitored but there exists a continuous decrease in the depth at which the dissolved oxygen goes to 1 mg/l inland from the inlet.

Dye Study

In order to obtain data on flushing in dead-end canals, a dye study was performed in Bayou Vista during July 1976. In the study, one gallon of 20-percent Rhodamine WT was added to the canal at station H-85 and mixed to an approximate depth of 4 feet (1.2 m). Dye concentrations were measured at 6, 12, 18, 24, 30, and 42 hours after dumping. Due to equipment limitations, no tidal measurements were taken during the study although the diurnal tides are generally less than 1 foot (0.3 m) (9). Dye concentration was monitored at seven stations along the canal with samples taken at top, 3-foot (0.9 m) and 6-foot (1.8 m) and 9-foot (2.4 m) depths.

At 10 hours after the study began, a brief thunderstorm occurred. Winds gusted to 35 miles per hour (56 m/hr) blowing parallel to the canal

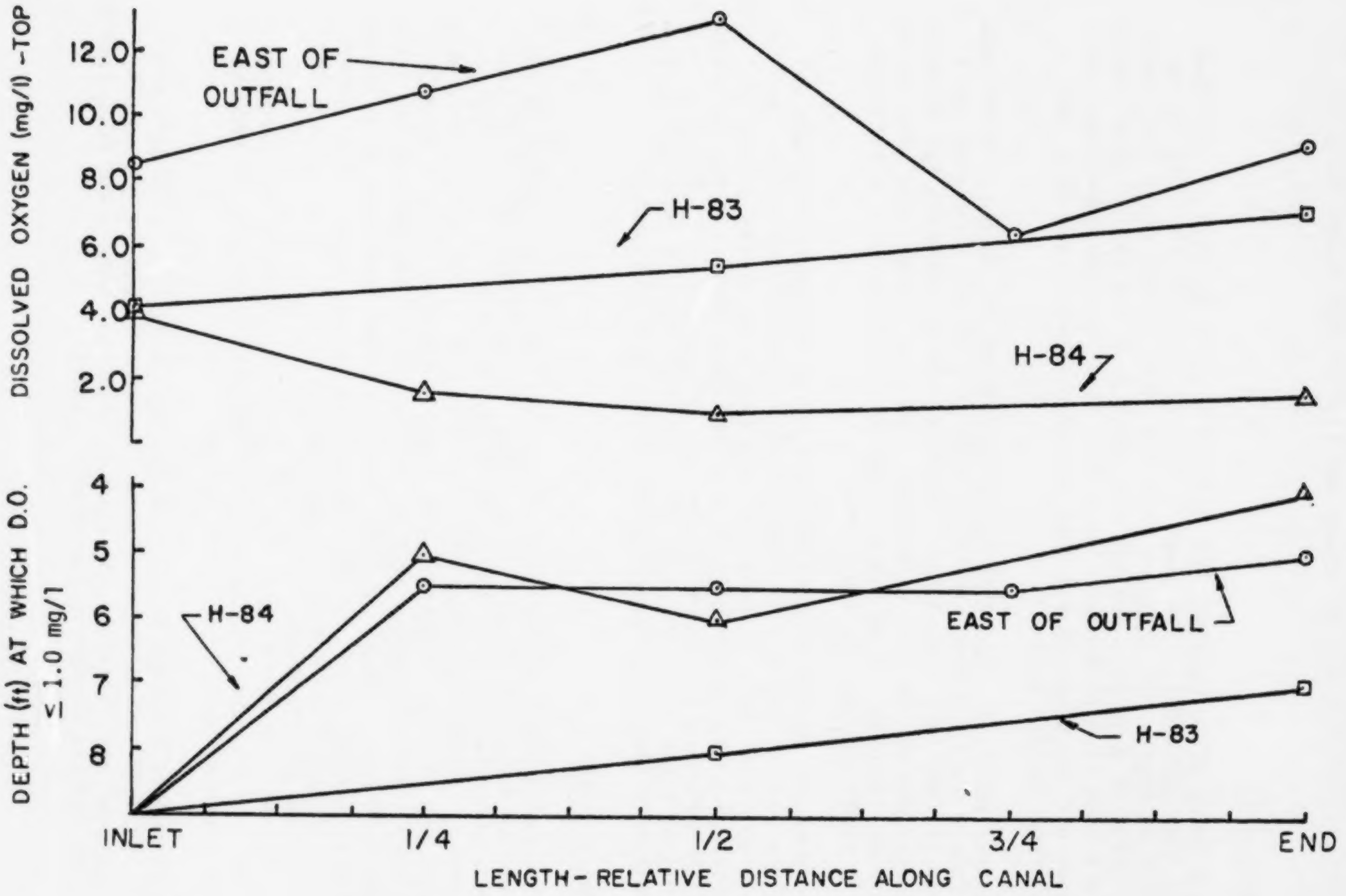


FIGURE 4. PROFILES OF THREE BAYOU VISTA CANALS EXAMINING VARIATIONS IN TOP D.O. AND DEPTH RELATIONS TO D.O. ≤ 1.0 mg/l

from the dead end toward the inlet. At this same time 0.3 inch (0.76 cm) of rain fell. The rain did not cause any significant runoff into the canals.

Canal H-85

At 6 hours after dumping, the dye had moved a maximum of 2000 feet (609.6 m) down the canal. The samples indicate good mixing down to the 6-foot (1.8 m) depth. At 12 hours from the start of the study (the storm event occurred at 9 hours), the dye had moved down the entire canal length down to a 6-foot (1.8 m) depth. The 9-foot (2.7 m) depth indicates some exchange of dye had occurred but that it had not moved past 1000 feet (304.8 m) from station H-85.

At this point, it should be noted that there is one basic difference between Canals H-83 and H-85. Station H-83 is a highly developed canal and nearly all the lots have homes. Station H-85 in contrast has approximately 15 percent of its lots with homes, and thus offers much less resistance to wind flushing. Wind flushing and mixing apparently contributed greatly to water movement in the canals.

At 18 hours, calculations indicate that 75 percent of the dye was still in the canal. The dye concentrations at the various depths tend to equalize with time. This is true with the exception of the 9-foot (2.7 m) depth, which remained independent of the shallower depths. This indicates poor mixing occurring somewhere between the 6- and 9-foot (1.8 and 2.8 m) depths. At 42 hours, approximately 60 percent of the dye remained in the canal. Figure 5 shows dye concentrations at 30 and 42 hours.

Jamaca Beach

The Jamaica Beach subdivision, shown in Figure 6 is located at the western end of Galveston Island on West Bay. It contains 1200 lots and approximately 600 houses with about one-third of the homes having waterfront footage. Runoff is collected in ditches and is discharged into West Bay outside the canals. The treatment plant discharges about 0.11 MGD (416.4 m^3/day) with an average BOD_5 of 7 mg/l and a suspended solids loading of about 10 mg/l. About 25 percent of the canal homes are on a sewage collection system and about 75 percent utilize septic tanks. Data shown in Figure 7 through Figure 8 are graphic comparisons of data between the two sections in Jamaica Beach. The canals range from 6 to 11 feet (1.8 to 3.4 m) in depth, with the 6-foot (1.8 m) depth occurring in the entrance canal. The shallow depth at the mouth of the system results in reduced mixing and flushing within the canal system.

Canal waters averaged 4.0 mg/l BOD_5, 18.1 mg/l TOC, 1.57 mg/l total nitrogen, and 0.29 mg/l orthophosphates. Nutrient levels in the runoff samples averaged approximately the same as the canal waters and BOD_5 levels averaged 160 percent higher in the runoff samples.

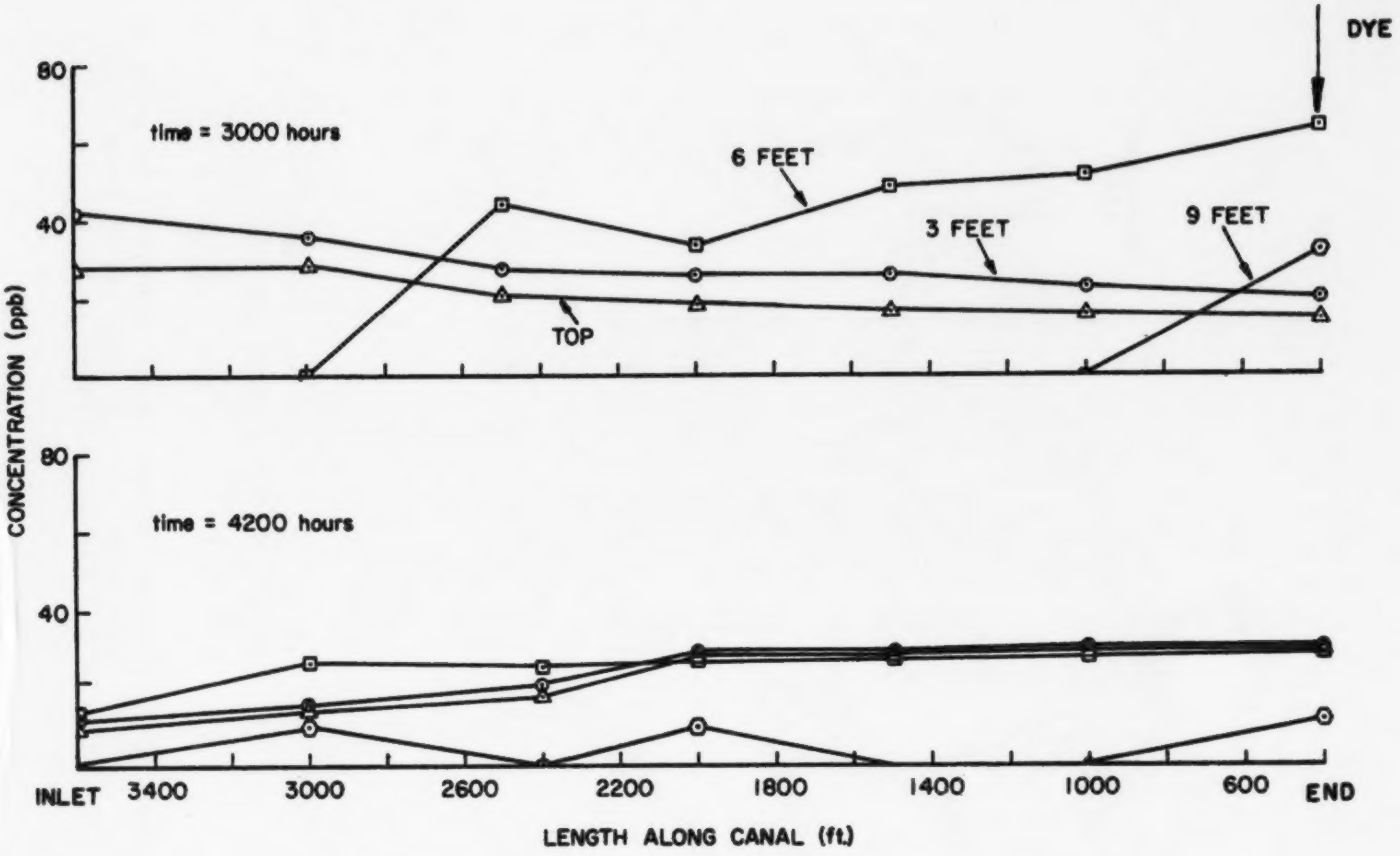


FIGURE 5. RODAMINE WT DYE STUDY CANAL H-85 (30-4200 hours)

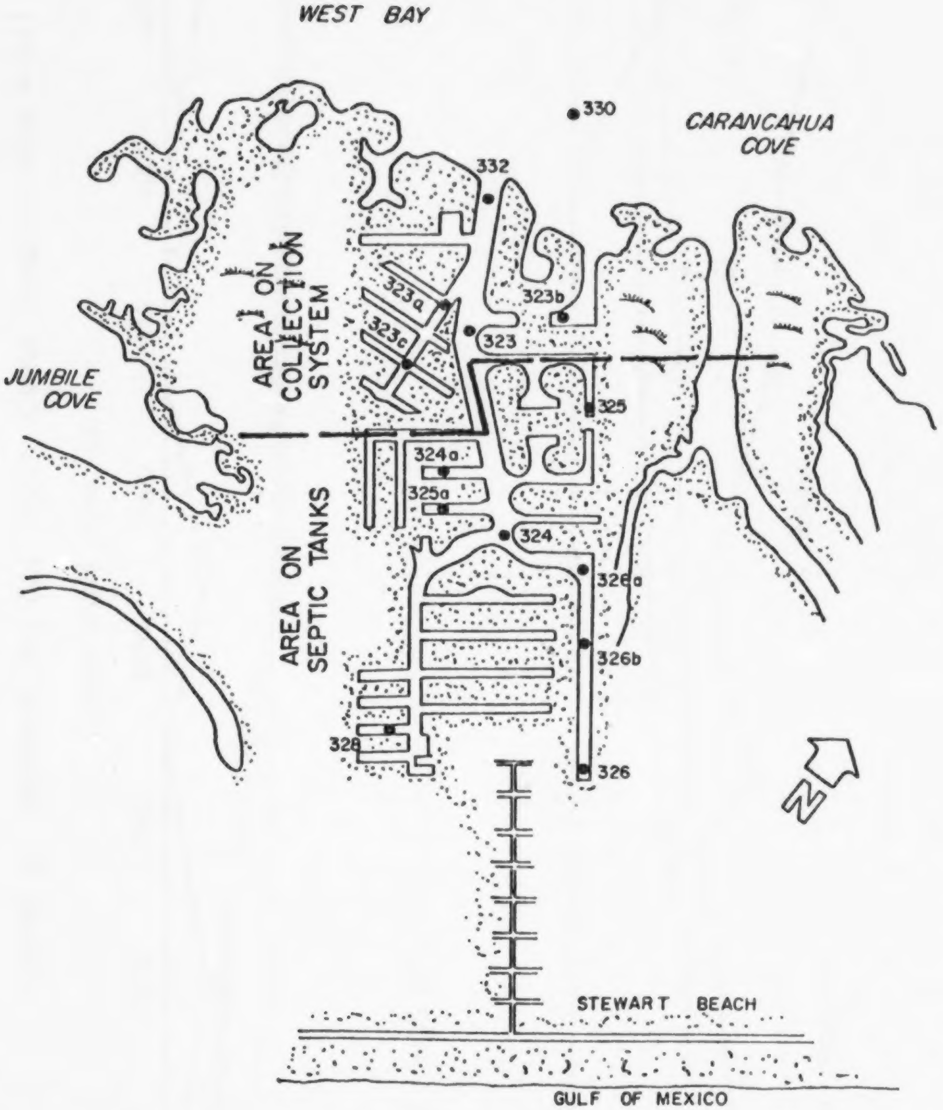


FIGURE 6. – JAMAICA BEACH CANAL COMMUNITY SHOWING COLLECTION SYSTEM AND SEPTIC TANK AREAS AND STATION LOCATIONS.

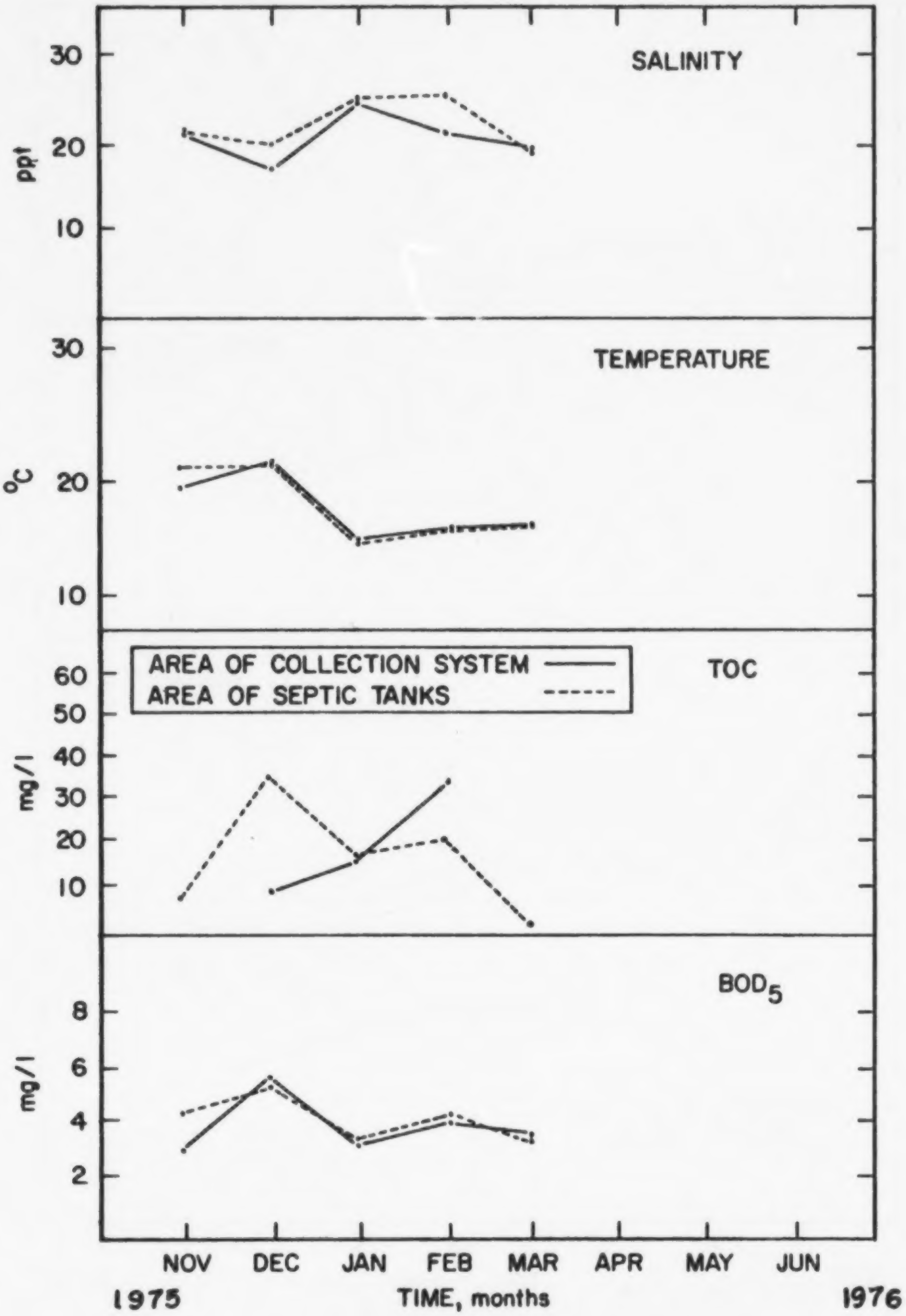


FIGURE 7. — COMPARISONS OF HYDROGRAPHIC VARIABLES BETWEEN AREAS OF A COLLECTION SYSTEM vs. SEPTIC TANKS IN JAMAICA BEACH

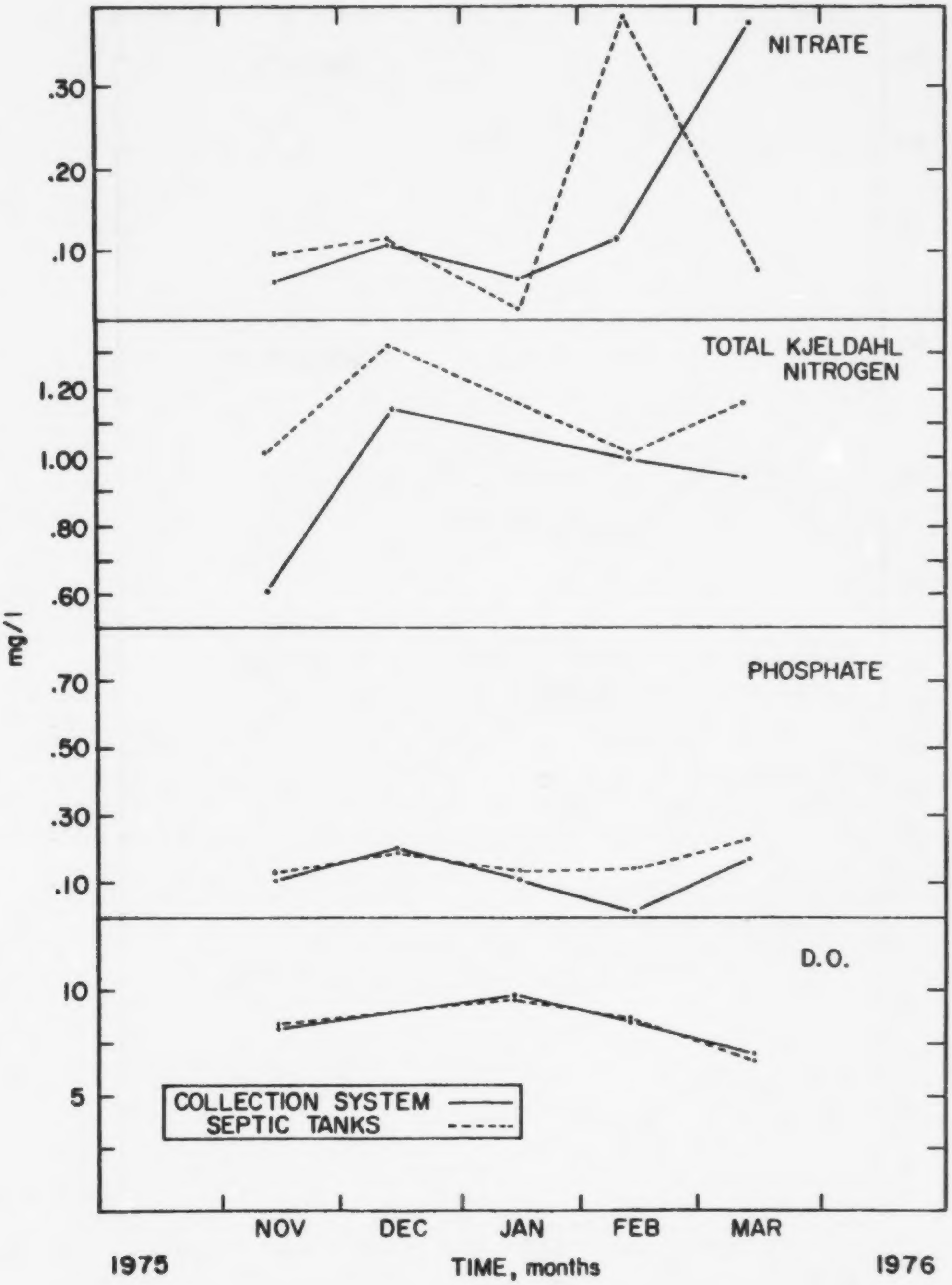


FIGURE 8. — COMPARISONS OF NUTRIENTS AND D.O. BETWEEN AREAS OF A COLLECTION SYSTEM vs. SEPTIC TANKS IN JAMAICA BEACH

Rainfall Runoff

Two methods of routing rainfall runoff are currently utilized in canal communities. One method is to allow runoff to flow overland into the canals without collection. This technique works well in canal communities because of the relatively short distances involved. The second approach is to collect runoff in open ditches or storm sewers and then discharge the runoff at points along the canal. The practice of collecting runoff intensifies water quality problems. Point discharge of runoff concentrates runoff pollution where allowing runoff to flow overland distributes the loading along the canals's entire length. Point discharge also increases the possibliity that sufficient energy will be imparted to the canal waters to resuspend bottom sediments. Resuspension of bottom sediments in effect increases pollution loading due to urban runoff.

D.O. Profile for Canal H-83 in Bayou Vista

As discussed previously, the Bayou Vista development in La Marque has had problems with several fish kills occurring in Canal H-83. Residents report that kills occur during the summer after rainstorms and at night. These conditions indicate kills resulting from depletion of oxygen in the canal waters.

In order to model the dissolved oxygen depletion that occurs during a rainfall runoff event, the equations and methods for estuarine analysis as presented by O'Conner and Eckenfelder (10) are applied to Canal H-83. Certain assumptions were made in applying this technique where data were inadequate or where the basic equations were not applicable. All calculations were made using USCG units, althougn SI units can be easily applied.

Runoff to be conisdered was from the area of land surrounding the upper half of the 3400-foot (1036 m) canal and rainfall was assumed to be 2.25 in/hr (5.72 cm/hr) which is the 2-year frequency. Application of the Rational Method for predicting runoff gives the runoff as Q = 5.26 cfs or 0.454 MGD (1718 m^3/day) for a time of concentration of 30 minutes, and a coefficient of runoff of 0.25.

At this flow rate the velocity, V, of the water in a canal 1050 ft^2 (97.5 m^2) cross-sectional area (100 ft wide and 10.5 ft deep) (30.5 m by 3.2 m) is equal to 18.0 ft/hr (5.49 m/hr). All four constants, that is K_2, E, J_1, and J_2, required to calculate the dissolved oxygen distribution were calculated (see thesis for detail (7)). The oxygen deficit equation is:

$$D = \frac{K_d}{K_2 - K_d} L_o \left(e^{-J_1 x} - e^{-J_2 x}\right) + D_o e^{J_2 x} \qquad (1)$$

where L_o is the BOD_u of the canal water, D_o is the initial oxygen deficit, and x is the distance from the point of discharge. Insertion of data gives,

$$D = \frac{0.980}{0.032 - 0.980} L_o(e^{-1.24x} - e^{-1.72x}) + D_o e^{-0.172x} \qquad (2)$$

Application of Dissolved Oxygen Sag Equation

The sag equation was applied to two cases. The first case simulates conditions in Bayou Vista where the sole source of BOD_5 is urban runoff. This case is equivalent to point discharge of urban runoff where bottom sediment is resuspended by the turbulence created by point discharge of the runoff.

Case I. Initial calculations were conducted using runoff BOD_u values collected during this study which averaged 14.3 mg/l (L_o) and D_o (initial D.O. concentration) which averaged 4.6 mg/l. Note that data used to determine L_o were not of first runoff and, therefore, the oxygen depletion resulting from application of these data is less than might be expected under actual conditions. The D.O. sag equation was applied using these data and an assumed critical dissolved oxygen level of 2 mg/l was reached in 30 hours. Under actual conditions, residents report kills occurring 12 to 15 hours after rainfall-runoff events; therefore, the model used does not describe the expected conditions accurately. Thus it became necessary to investigate further into the problem to account for sediment resuspension.

Case II. Previous work by Hall (8) on the canal sediments indicates a strong possibility that the point discharge of runoff into the canals is sufficient to cause sediment resuspension. In order to investigate this possibility, a hydraulic model of Canal H-83 was designed and constructed.

The width and depth of the model were 1/25 that of the actual canal. The model was based on providing a plug-flow travel time of 12 hours. The model measured 4 ft (1.2 m) by 6.5 ft (2.0 m), respectively.

In order to simulate the scouring action of runoff in the canal sediments, the Reynolds Number was used in the form Re=Vd/v, where V is the velocity in ft/sec, d is the characteristic depth, and v is the viscosity. The characteristics depth, d, used was sediment particle diameter. Since sediment from Canal H-83 was used in the model, partical diameter, d, was fixed. In addition, viscosity, v , was constant from model to canal for a given Reynolds Number. Field measurements indicated that the gravity head for runoff discharge into the canals was about 2.5 ft (0.8 m). The resulting velocity was 12.5 ft/sec (3.8 m/sec) which was used in the model.

The last scale-down required was the calculation of runoff flowrate. This was done by determining the energy per unit volume imparted by runoff. The energy is equal to mass flow per unit time multiplied by the elevation head. The runoff for one quarter of the drainage area which discharges runoff into the canal was used to calculate mass flow per unit time. Since the finished grade is about 2.5 ft (0.8 m) above the canal water surface, an elevation head of 2.5 ft (0.8 m) was used.

The scale-down factor for energy can, therefore, be determined simply by the ratio of model volume over canal volume. The flow rate is then calculated by dividing the scaled-down power by elevation head and converting mass flow rate to volumetric flow rate. The resulting volumetric flow rate was 210 ml/min. The discharge outlet was then sized to provide 210 ml/min at 12.5 ft/sec (3.8 m/sec).

The model was built to operate according to the criteria established above. Concrete was used to form the shape of the canal bottom and the canal sediment was distributed evenly over the bottom with a depth of about 1 inch (2.54 cm). The model was then filled with water and left for one day to allow suspended solids to settle.

To run a test, the pump which provided the runoff flow was started and samples were collected of the water close to the point of discharge. The samples were then analyzed for suspended solids.

Work by Hall (8) on the canal sediments was then used to convert the suspended solids data to BOD_u. According to Hall, the maximum oxygen demand for Bayou Vista sediments was 17.94 mg O_2/g sediment. Suspended solids data collected from the model had an average value of 1.25 g/l suspended solids. Therefore, if 1.25 g/l of bottom sediment are suspended by runoff, the oxygen demand, L_o, of the suspended is the initial oxygen demand of the canal water, 14.3 mg/l, plus the oxygen demand of the sediment.

$$\begin{aligned} L_o &= (17.94 \text{ mg/g})(1.25 \text{ g/l}) + 14.3 \text{ mg/l} \\ &= 36.7 \text{ mg/l} \end{aligned}$$

After the model testing, all variables and constants required to solve the sag equation, Equation 2, were known. Figure 9 is a summary of results from application of the sag equation and indicates good comparability with conditions reported by residents. The critical oxygen deficit of 2 mg/l was reached in 11.4 hours versus the 12 to 15 hours reported under actual conditions.

It should be noted that application of the D.O. sag equation requires steady state conditions. Therefore, strict application of the equation required that runoff be assumed to be constant over the required application period.

This is not, however, an unreasonable assumption. Although rainfall intensity decreases with duration, the rate of runoff decreases more slowly with time due to a corresponding decrease in the infiltration rate. Calculations for the Galveston area indicate that at 11.4 hours the rate of runoff for a 2-year frequency rainfall event will be 65 percent of the runoff

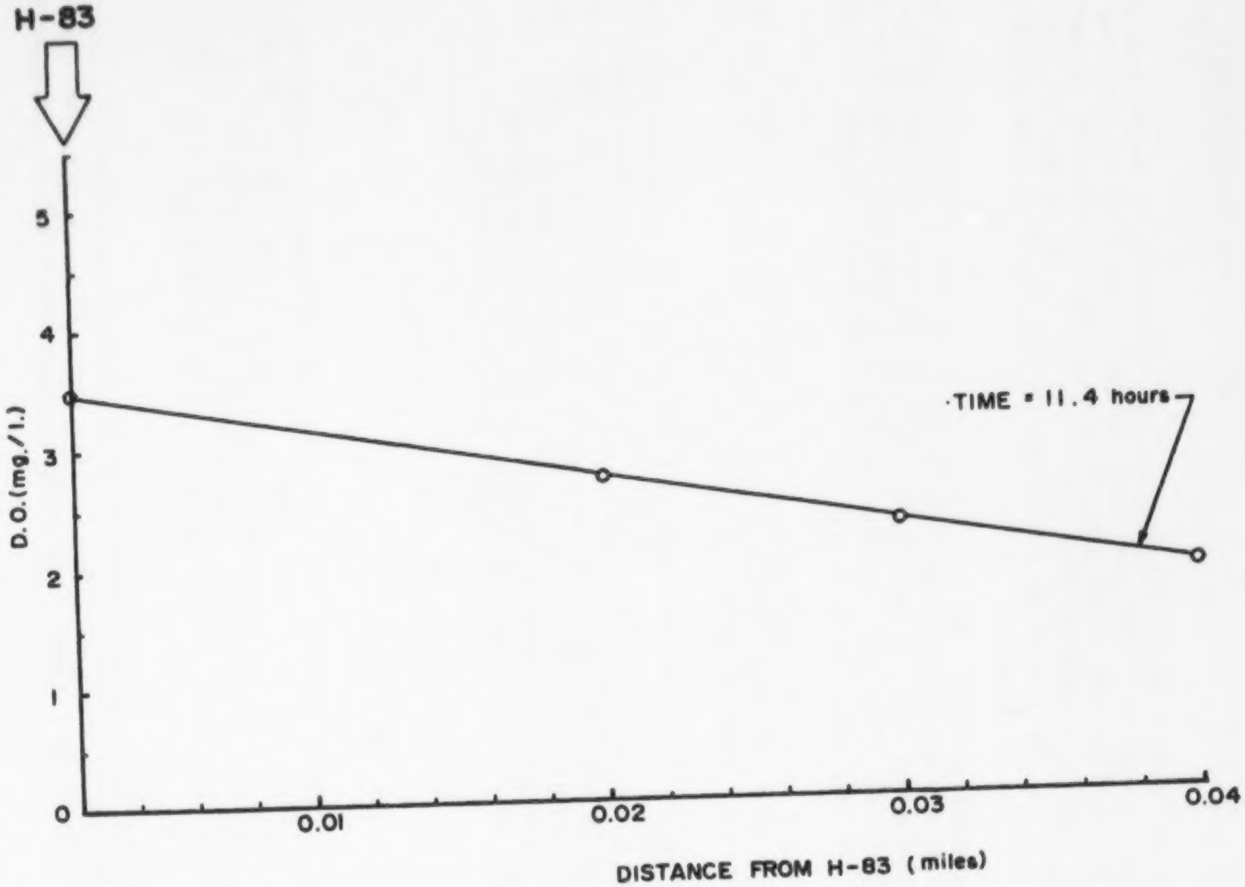


FIGURE 9. D.O. SAG PROFILE — CANAL H-83.

rate at the minimum time of concentration. Thus, application of the modified Streeter-Phelps sag equation in the manner described approximates actual conditions encountered in the study area.

CONCLUSIONS

1. Septic tanks are not a suitable form of domestic waste treatment in canal communities based on coliform levels in the septic tank area of Jamaica Beach (7,8).

2. Domestic wastewater should be collected and treated at a community wastewater treatment plant. The discharge from the plant should be located so that it does not pass into the canal system.

3. Water quality in canal communities is strongly influenced by the circulation of the surrounding natural waters through the system. To enhance circulation, the following criteria should be observed.

   a) Canals should be oriented to maximize wind flushing, i.e., aligned with the prevailing winds.

   b) Canals should not be dredged any deeper than is required to allow safe boating. In most cases an 8-foot (2.4 m) depth will be sufficient. If at all possible, the canal depth should not be greater than the parent body of water.

   c) Complex canal systems with restrictive inlets should be avoided.

   d) Designs which provide for circulation of the surrounding waters through the canals should be encouraged.

   e) The canals should be of uniform depth, however is possible, the bottom should gradually slope from the furtherest point inland to the parent body of water. This would help in preventing stagnent water pockets.

4. If the canals are designed with sufficient circulation, the runoff should not be collected. Instead, the roads and lots should be designed with sufficient slope to remove runoff by overland flow into the canals. Point discharge of runoff concentrates pollution and may cause sediment to be resuspended from benthic deposits. This design will not create the mosquito breeding problems of an open ditch collection system.

5. If runoff is collected for point discharge, drop boxes should be used to dissipate excess energy prior to release into the canals in order to prevent suspension of benthic sediments.

6. Dead-end systems should be avoided if possible. When used, dead-end canals should be connected by culverts to enhance circulation. Dead-end canals of 3400 feet (1036.3 m) have had fish kills occur due to oxygen depletion.

7. Storm-water runoff into the canals should be minimized as much as possible and runoff from adjacent areas should not be discharged within a canalled community.

8. Concrete bulkheads should be used instead of timber. Timber bulkheads are susceptible to collapsing which allows soil to enter the canal and create stagnent water pockets.

REFERENCES

1. Goyal, S. M., et al., "Prevalence of Human Enteric Viruses in Coastal Canal Communities." Journal Water Poll. Control Fed., 50, 10 (1978).

2. Gloyna, E. F., Malina, J. F., Jr. "Galveston Bay Water Quality Study, Historical and Recent Data." The Texas Water Pollution Control Board. Austin, Texas (March 1964).

3. Lindall, W. N., Jr. and Trent, L. "Housing Developments in the Coastal Zone of the Gulf of Mexico: Ecological Consequences, Regulations, and Recommendations." MFR Paper 1163 (1976).

4. Taylor, John L., and Saloman, C. H. "Some Effects of Hydraulic Dredging and Coastal Development in Boca Ciega Bay, Florida." Fishery Bulletin 67:213-214 (1968).

5. Moore, D. and Trent, L. "Setting, Growth and Mortality of Crassostrea virginica in a Natural Marsh and a Marsh Altered by a Housing Development." National Shellfisheries Assoc., Vol. 61 (June 1971).

6. Paulson, O. L. and Pessoney, G. F. "Residential Canals along the Gulf Coast." National Technical Information Service, PB-243-779 (1975).

7. Messenger, A. L., "Water Quality and Urban Runoff in Selected Canal Communities along the Texas Coast." Thesis,Texas A&M Univ. (1979).

8. Hall, Ernest. "Water Quality of Some Coastal Canal Communities in the Galveston County Area." Thesis, Texas A&M Univ. (1976).

9. United States Coast Pilot 5 Atlantic Gulf Coast of Mexico, Puerto Rico and Virgin Islands." 9th Ed., NOAA Nat. Ocean Survey.(July 1976).

10. Eckenfelder, W. W., Jr. and O'Conner, D. J. Biological Waste Treatment. Pergamon Press. New York, New York (1961).

7

Sixth Session

ECOLOGICAL RESPONSE TO STORMWATER

Moderator: James S. Taylor
University of Central Florida
Orlando, Florida

THE RESPONSE OF SUBTIDAL INFAUNAL COMMUNITIES TO A CHANGE IN WASTEWATER DISCHARGE [1]

Ronald M. Thom and Kenneth K. Chew
College of Fisheries
University of Washington
Seattle, Washington 98195

ABSTRACT

Certain aspects of the structure of macroinfaunal communities in the subtidal sediments adjacent to an intertidal combined sewage over (CSO) were studied during 1978. The impact of this ephemeral source of raw wastewater and street runoff on the communities was evaluated using samples taken after periods of highest (April) and lowest (August) frequency of discharge. In April, the community nearest the overflow was characterized by a high number of individuals, a low number of taxa, a low species diversity, a high abundance of the polychaete Capitella capitata, and a high relative number of polychaetes. Subsurface deposit feeding species dominated the community. There was an area of markedly low infaunal abundance bordering the region of acute impact. Diversity and number of taxa were highest at the sites furthest (i.e. > 1000 m) from the overflow. Bivalve molluscs were in relatively high abundance, and carnivores, surface deposit feeding and subsurface deposit feeding species were approximately equally abundant at these sites.

The samples in August showed similar trends relative to the overflow in number of individuals, number of taxa, species diversity and community composition by species, phyla, and feeding type. However, differences among all sites in these parameters were less pronounced during this period. A notable between season difference was the switch in numerical dominance from C. capitata to the leptostracan crustacean Nebalia pugettensis at the site nearest the overflow. This switch may be related to decreased disturbance by effluent flows near this site prior to the August sampling.

A cluster analysis indicated that depth was of primary importance, and that the CSO was of secondary importance in producing among site differences. The communities at the shallowest sites and northward from the overflow were the most altered by the effluent.

We concluded that the decreased frequency of overflows in August may be responsible for the relative decrease in differences among the communities at all distances from the overflow. Impact studies should be conducted over

---

[1] Contribution No.   from the College of Fisheries, University of Washington, Seattle, Washington 98195

several seasons in areas where substantial variations in wastewater and/or natural environmental parameters occur.

INTRODUCTION

Most of the sewers in the city of Seattle, Washington, carry both sewage and street runoff. During rainy periods, sewage treatment plants become overloaded and some of the material in the sewers is released through combined sewer overflows (CSO). The CSO discharging the greatest volume of sewage into marine waters in Seattle is located at Denny Way in Elliott Bay, Puget Sound (Fig. 1). Overflows through this CSO occur approximately 38 times a year with an average volume per overflow of 2.8 x 10^6 gal (10.6 x 10^6 l) (Municipality of Metropolitan Seattle - METRO, unpublished data). The overflows are most frequent, of greater volume, and have the highest flow rates during winter and early spring.

Effluent is discharged through a conduit located within a small cove in the littoral zone. The conduit is surrounded by a steeply sloping boulder sea wall, which extends approximately 200 m south and 1400 m north from this point. The path of the effluent is marked by a scoured zone in the soft sediment which forms the substrata in the cove directly seaward from the base of the conduit.

Armstrong *et al*. (1978, 1980) studied the impact of this source of pollution on the adjacent intertidal and subtidal epibiotic and infaunal communities during spring (ie. April-May). They documented significant changes in the structure of the communities, and these were measured as changes in species richness and diversity, community composition, and trophic requirements. Armstrong *et al*. (1980) were unable to locate in the literature similar studies near CSO's, but did conclude that the modifications they saw were in line with those reported for areas subjected to continuously flowing effluent from intertidal and subtidal outfalls (eg. Reish 1959, Littler and Murray 1975, Stevenson *et al*. 1975, Smith and Green 1976). Armstrong *et al*.(1978) recommended that, due to the variable nature of the CSO discharge, studies be conducted during several seasons to document any seasonal aspects of the CSO impact on the biological communities.

The purpose of the present paper is to compare the structure of subtidal infaunal communities near the Denny Way SCO in an extreme case; that is, shortly after periods of maximum and minimum number of overflows. Therefore, we sampled 14 subtidal sites during April and August, 1978, and followed the methods of Armstrong, *et al*. (1978) for sample processing and data analysis.

MATERIALS AND METHODS

Sampling sites were located at 9 and 13 m depth contours along seven transects which ran approximately perpendicular to the shoreline. Transects 1, 2, 3, 4, 6 and 7 were positioned 1270, 320, 260, and 180 m north and 150 and 440 m south of the CSO, respectively. Transect 5 was located immediately seaward from the CSO.

Two Van Veen (0.1 m^2) grab samples were taken at each site using the R/V HYDOR. The texture, odor and color of the sediment was described, and a small (3-5 g) subsample was extracted for volatile organics analysis. The

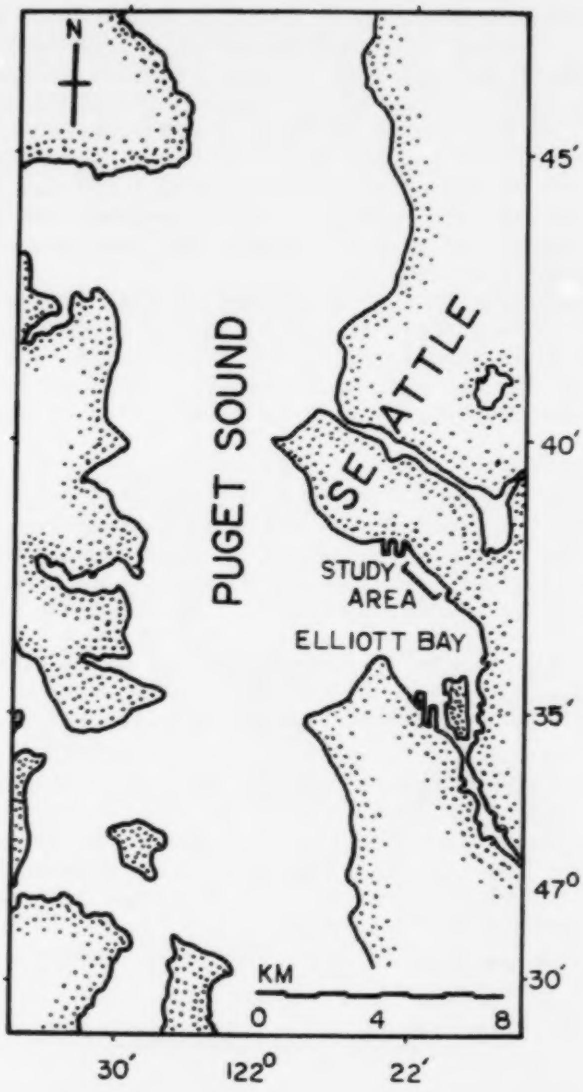


Fig. 1. The location of the study area.

samples were live screened with gentle shaking in water through a 1 mm mesh sieve on board. The material retained on the sieve was placed into glass jars and preserved and stained in 10% formalin and Rose Bengal stain. In the laboratory, the samples were rescreened (1 mm mesh) and rinsed with tap water to remove the formalin. All organisms retained on the screen were identified (usually to species) and counted.

Volatile organic content of the sediment was determined by drying the 3-5g subsample for 30 hours at 100° C, recording the weight, burning the material for 3 hours at 550° C, and reweighing. The loss of weight, expressed as a percentage of the dry weight, gives a rough estimate of the organic carbon content of the sediment (Morgans 1956).

Species diversity at each site each season was estimated using the combined data from the replicate samples. The measure of Hurlbert (1971) was used for this estimate, and it is calculated by the formula

$$E(S_n) = \sum_i \left[1 - \frac{\binom{N-N_i}{n}}{\binom{N}{n}} \right]$$

= expected number of species in a sample of n individuals selected randomly from a collection of N individuals, S species, and N_i individuals in the ith species.

Curves for the two most abundant taxocenes (ie. polychaetes, molluscs) relating $E(S_n)$ to several values of n were constructed for each site each season.

Species composition at the sites was compared using classification analysis (Boesch 1977). The species abundances at each site were square root transformed. Intersite dissimilarities were determined using the Bray-Curtis coefficient of dissimilarity (Clifford and Stevenson 1975). The sites were then clustered using the group average sorting strategy (Sokal and Michener 1958). The results of the classification analysis are displayed as a dendrogram.

RESULTS

During the three months prior to the April sampling, 25 overflows occurred (total volume = 275 x 10^6 l), with an average flow rate of 45 x 10^3 l/min. Three months prior to the August sampling, five overflows took place (total volume = 45 x 10^6 l) at an average rate of 34 x 10^3 l/min.

The sediment collected in the grab samples graded from light brown sand at transect 1 to fine black silt at transect 5 (Table 1). Sulphide odor and volatile organic content were generally highest in sediment at the sites nearest the CSO (Table 1). Sulphide levels generally decreased, and sediment color lightened, at the sites south of the CSO. In April, black sediment with a moderate sulphide odor was encountered at the 9 m site on transect 2, which is located relatively far from the CSO. This site is in front of a small cove, and this cove may contribute to the development of eddy currents and the deposition of suspended material.

Infaunal densities differed appreciably among the samples from 9 m in

Table 1. Characteristics of Subtidal Sediments Sampled Near the Denny Way CSO.

Date	Transect	Depth (m)	Texture	Color	H_2S Odor	Vol.Org.(%)
April	1	9	sandy	lt. brown	none	1.5
		13	"	"	"	0.8
	2	9	fine silt	black	moderate	2.0
		13	"	brown	slight	2.2
	3	9	"	lt.brown/black	"	2.4
		13	"	"	"	2.3
	4	9	"	"	"	11.7
		13	"	"	"	2.4
	5	9	"	black	moderate	3.3
		13	"	"	"	3.3
	6	9	"	lt.brown/black	slight	2.6
		13	"	"	"	7.8
	7	9	"	"	"	2.8
		13	"	"	none	2.2
August	1	9	sandy	brown	none	1.1
		13	"	"	"	2.3
	2	9	fine silt	green-black	slight	2.1
		13	"	"	"	1.8
	3	9	"	"	"	-
		13	"	"	"	2.7
	4	9	"	"	"	1.9
		13	"	"	"	2.6
	5	9	"	black	strong	2.6
		13	"	"	"	5.5
	6	9	"	lt.brown/black	slight	2.2
		13	"	"	"	1.4
	7	9	"	"	"	3.2
		13	"	"	"	2.4

April (Fig. 2a). The pollution indicating polychaete Capitella capitata accounted for the majority (76%) of the individuals found at the site nearest the CSO. Total density generally declined at sites immediately north and south of the CSO, and increased at the sites furthest from the CSO. This trend was also seen in August, although abundances near the CSO were not as high as in April. C. capitata was in low abundance at all sites in August, and the crustacean Nebalia pugettensis comprised the majority (52%) of the individuals on transect 5. Density values were notably low at the 9 m site on transect 2 during both seasons.

The highest densities along the 13 m contour in April occurred at sites on the transects immediately north (4) and two transects south (7) of the CSO (Fig. 2b). The bivalve mollusc Axinopsida serricata was numerically dominant (44 and 59% of the total number of individuals on transects 4 and 7 respectively) at these sites. Very few (total=15) individuals of C. capitata were found in the samples from 13 m.

Density values for the samples taken at 13 m in August were substantially different from those in April (Fig. 2b). Relatively low densities occurred at the sites on transects 4 and 7, where values decreased by approximately 72 and 58% respectively. The highest densities occurred far from the CSO, and these values were approximately double those noted in April at these sites. The site on transect 5 showed the least change in this parameter.

The number of taxa found at both the 9 m (Fig. 2c) and 13 m (Fig. 2d) differed among sites, and exhibited a general decline at decreasing distance from the CSO. An exception to this was the samples from the site nearest the CSO. The higher number of taxa at this site may be related to the higher number of individuals that were collected. These data differed between seasons, although the trends were similar. The flux in number of taxa between samplings was most pronounced on transects 5 and 6 at 9 m, and on transect 4 at 13 m. The greatest number of taxa were consistantly found in the samples from transect 1.

Mollusc species diversity (as measured by curve height) was lowest at the sites nearest, and immediately south of, the CSO at 9 m in April (Fig. 3a). Mollusc diversity was correlated positively with distance from the CSO at the 13 m contour during this season (Fig. 3b). In August, mollusc diversity at 9 m was similar among all sites (Fig. 3c). Lowest diversities were encountered at sites nearest the CSO at 13 m during August.

Polychaete species diversity decreased with decreasing distance from the CSO at 9 m in April (Fig. 4a), and showed a similar but less pronounced trend at 13 m. Several sites at 9 m grouped with the site closest to the CSO in August, which indicated similarities in polychaete diversity (Fig. 3c). The site furthest from the CSO was again highest in this parameter. A gradient relative to the CSO may be indicated at 13 m by the relatively low diversity at the site on transect 4 and the relatively high diversity at the site on transect 1 (Fig. 4d).

The classification of site samples taken in April revealed the primary importance of depth in controlling the species composition, and the secondary but substantial effect the CSO effluent had on the communities at some sites (Fig. 5a). Depth appeared to be not as important in August, however the influence of the CSO was still evident during this month (Fig. 5b). Five subgroups were designated by letters, and these letters were plotted on maps of the sampling locations (Figs. 6a, b). If transect 1 can be regarded as a

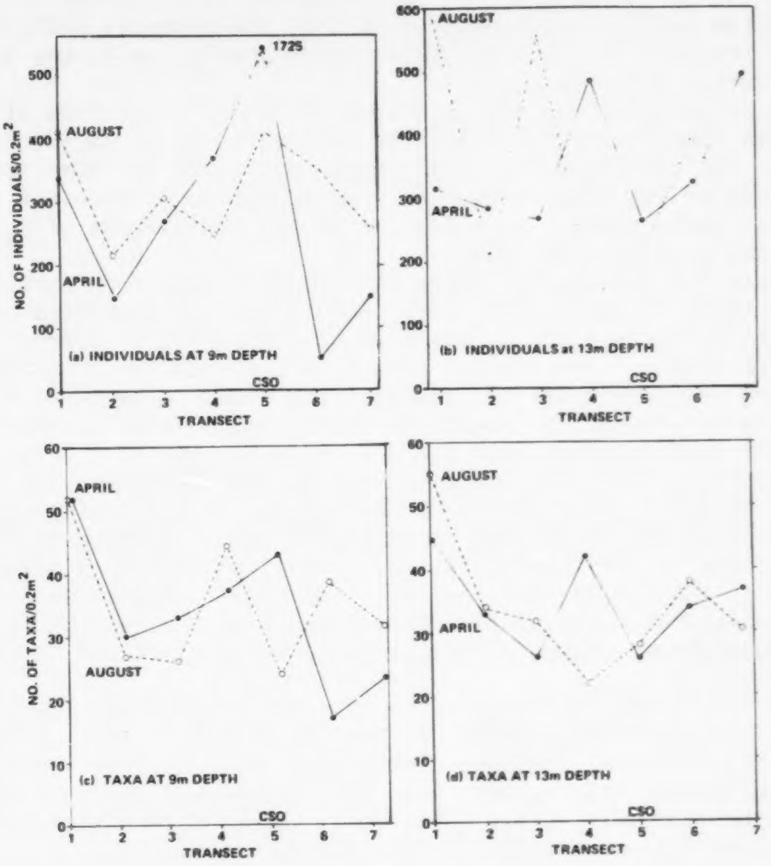


Fig. 2. Density of individuals and taxa found in the subtidal samples from near the Denny Way Regulator outfall during April and August, 1978.

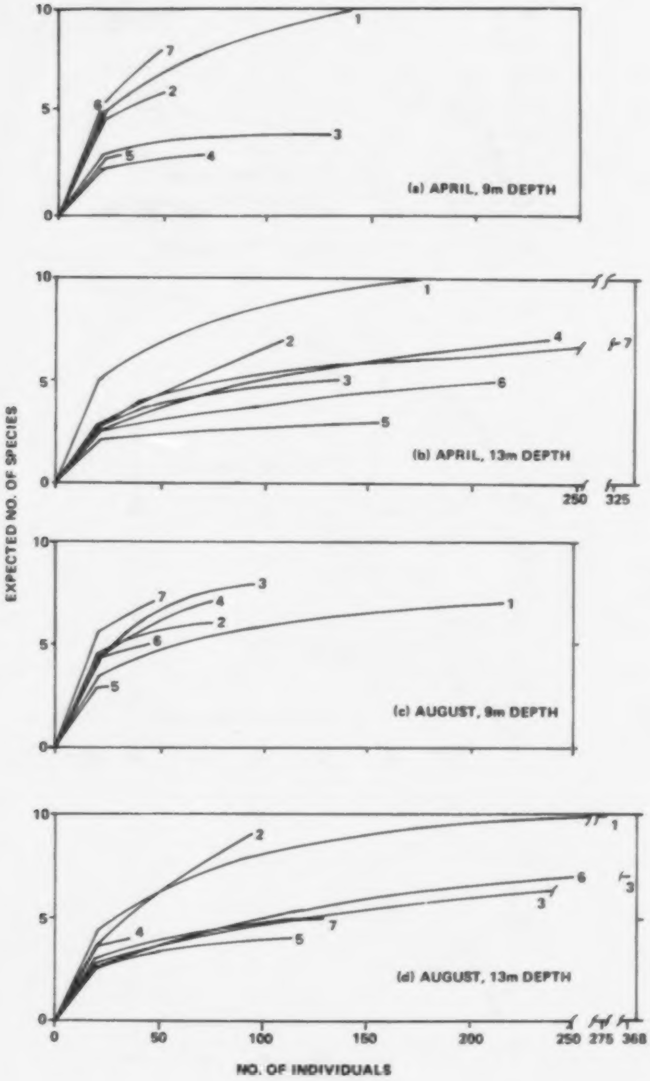


Fig. 3. Species-richness curves for molluscs collected near the Denny Way Regulator outfall during April and August, 1978. The numbers indicate sampling transects.

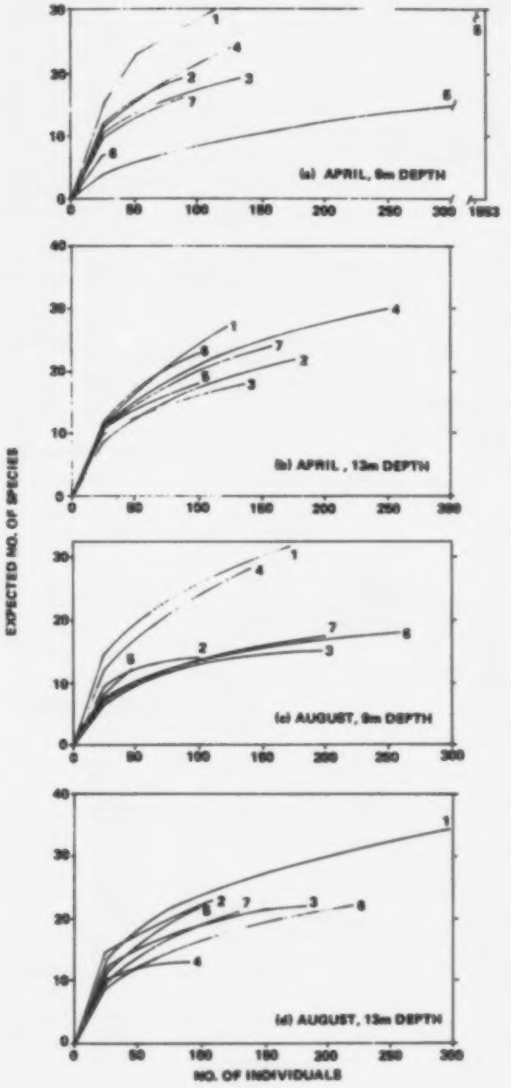


Fig. 4. Species-richness curves for polychaetes collected near the Denny Way Regulator outfall during April and August, 1978. The numbers indicate sampling transects.

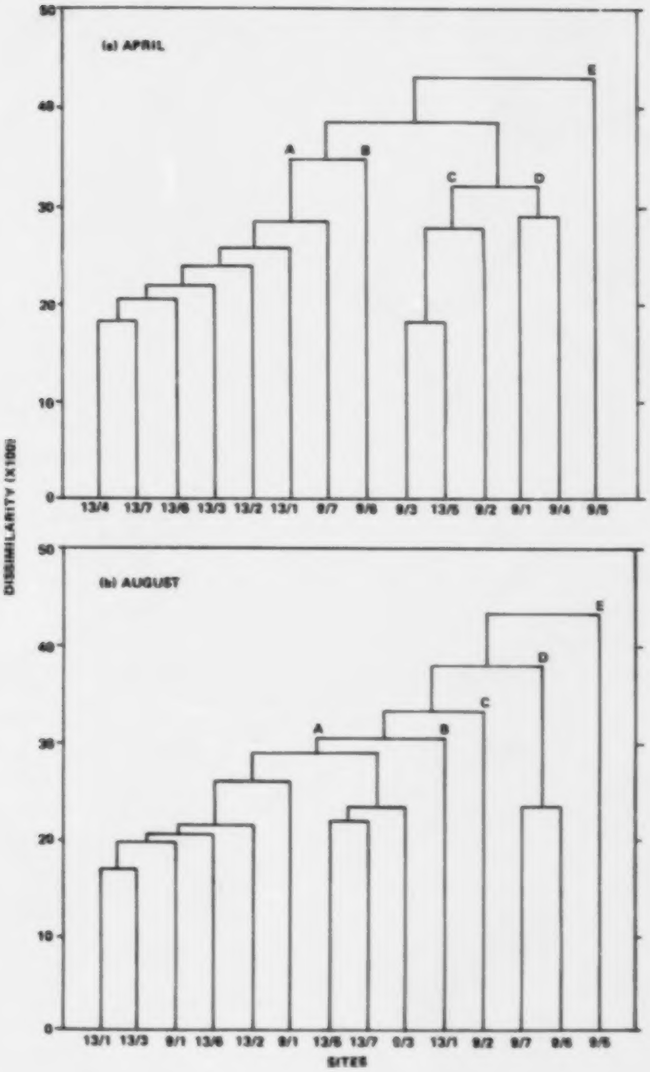


Fig. 5. Dendrogram of subtidal infauna samples from near the Denny Way Regulator outfall, 1978. The site designations are given as depth (m)/transect.

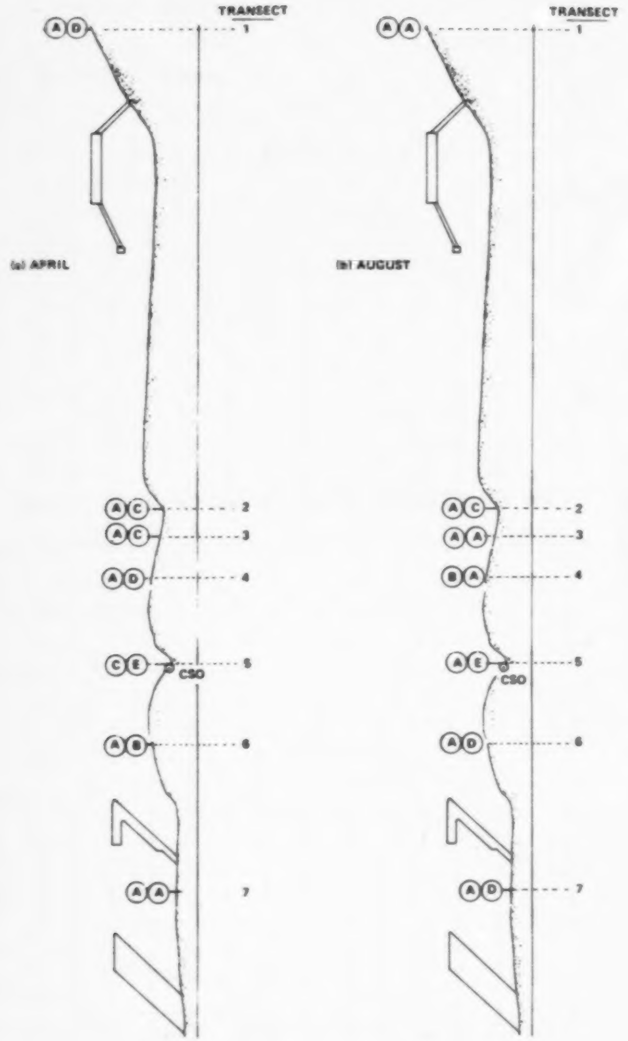


Fig. 6. Positions of subgroups of sites from cluster analysis of samples of subtidal infauna collected near the Denny Way Regulator outfall, 1978.

control, then modifications caused by the CSO effluent and evident in April are seen at sites 9/2, 9/3, 9/5, 13/5, and 9/6. The gradient of influence at depth (13/5) and distance (9/2, 9/3) is apparent. Interestingly, the community along transect 4 appears unmodified.

August samples revealed that the CSO affected infaunal community composition at some sites different than those affected in April (Fig. 6b). Sites 9/5, 9/2 and 9/6 were still modified, and sites 13/4 and 9/7 appreared to also be influenced. Lower flow rates prior to the August sampling coupled with natural northward flowing currents may have resulted in the effluent being carried to the 13 m site on transect 4 rather than affecting the 13 m site on transect 5.

A graphic display (after Snee 1974) of the proportions of individuals among three major phyla found at each site provides a partial indication of the influence of the CSO on the benthos (Fig. 7a). Infauna composition at 9 m on transect 5 was significantly different ($P < 0.05$) from that at the other sites during both April and August. The April and August samplings at this site were also significantly different from one another. Most of the individuals in April were polychaetes. Conversely, a large proportion of crustacea comprised the samples from this site in August. The samples taken at 9 m on transect 1 were significantly different from one another, but they did exhibit an infaunal composition that was relatively higher in mollusc individuals and lower in numbers of polychaete individuals. The sites closest to the CSO had increased ratios of polychaetes to molluscs.

A change between seasons in these data from 9 m was evident. Samples from the same site were significantly different between samplings except at transect 2. The greatest difference (i.e. as measured by distance between points on the graph) is between samples at sites on transects 4, 5, and 6. Considering transect 1 a control area, where seasonal changes can be regarded as natural, it can be concluded that the effluent from the CSO tends to increase between-season differences in the infauna at 9 m.

The graphical analysis of site from 13 m revealed that there were no significant differences among sites or between seasons at a site (Fig. 7b). However, the August samples from transect 4 were disproportionately high in polychaetes which may suggest a possible influence of the CSO at this depth.

Polychaete feeding strategies are a reflection of the characteristics of the sediment and overlaying water (Jumars and Fauchald 1976). Burrowing (subsurface) deposit-feeding polychaetes were proportionately most abundant near the CSO (Fig. 8a). At the site at 9 m on transect 1, the three major feeding types were in approximately equal abundance (Fig. 8a). These results were true for both the April and August samples; however, the differences in proportion of feeding types among sites were generally greatest during April. Significant between season differences were seen at sites on transects 4, 5, 6, and 7. Samples at sites on transect 6 and 7 in August contained large numbers of the surface deposit-feeding polychaete Prionospio steenstrupi which accounted for the position of these samples on Fig. 8a. At 13 m, there were no significant differences among samples (Fig. 8b). The samples from April tended to have proportionally more burrowing deposit feeders, however.

DISCUSSION

Effluent from the Denny Way CSO significantly affects the adjacent shallow subtidal infaunal communities, and some aspects of this effect changed with season. Frequent overflows of high volume and rate resulted in an increase

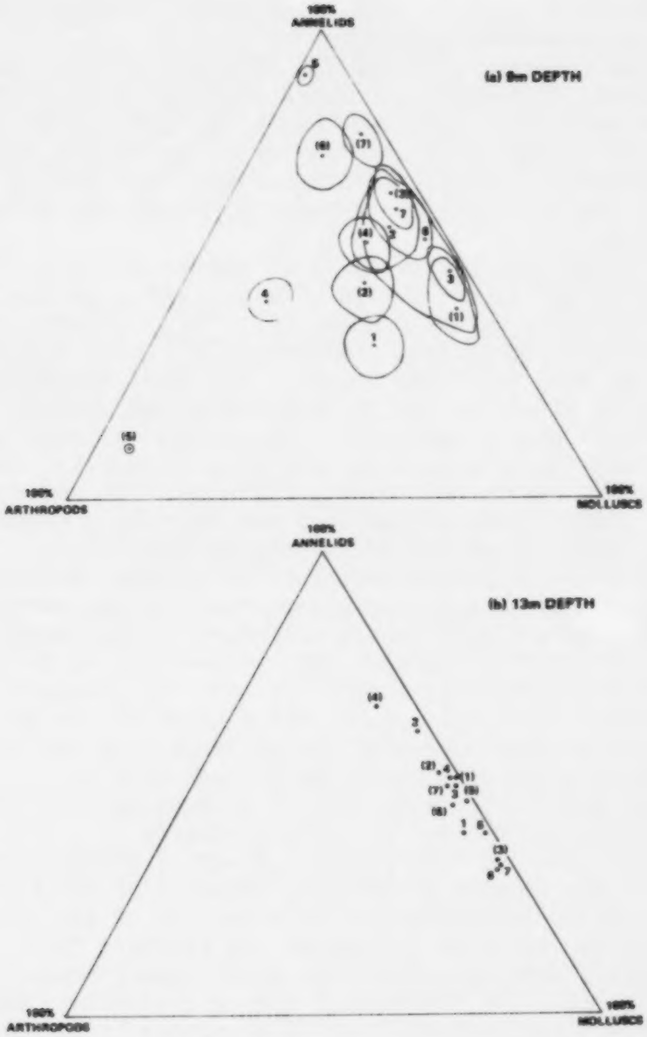


Fig. 7. Proportions of the total numbers of annelids, arthropods and molluscs sampled near the Denny Way Regulator outfall, April and August, 1978. The numbers designate transects; those in parentheses are for August. The circles indicate 95% confidence intervals.

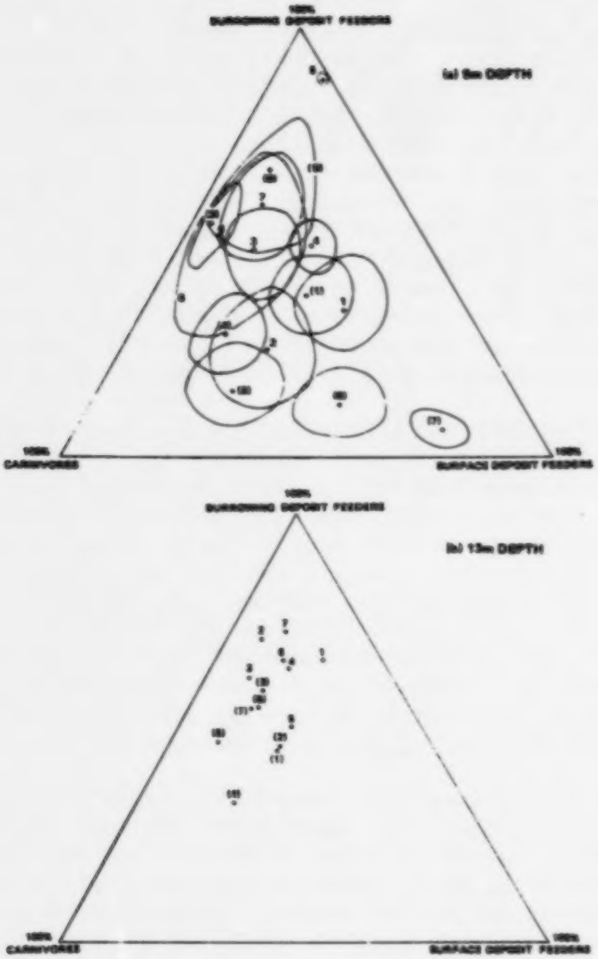


Fig. 8. Proportions of total numbers of polychaetes within each of three feeding-type categories, for samples collected near the Denny Way Regulator outfall, April and August, 1978. The numbers designate transects; those in parentheses are for August. The circles indicate 95% confidence intervals.

in infaunal density, an increase in the abundance of the pollution indicating polychaete Capitella capitata, reduced mollusc and polychaete diversities, raised relative proportions of annelids as compared to other phyla, and a predominance of burrowing deposit feeding annelids at the site closest to the CSO. The effect was reduced, but evident, at sites immediately beyond this area and at some sites to the north. Data on sediment quality corresponded with these results, which suggest that the CSO effluent is responsible for the modified conditions near the discharge.

After a period of infrequent overflows, the differences in community parameters among sites were less pronounced. The greatest change in the community between seasons occurred at the site closest to the CSO. Mollusc and polychaete species diversities remained low; however these values resembled those for sites immediately north and south of the CSO. Burrowing deposit feeding annelids were proportionally most abundant at 9 m sites (on transect 3 and 5) near the CSO, and the proportions of annelid feeding types changed significantly between seasons at all sites except those on transects 1, 2, and 3.

In August, the leptostracan crustacean Nebalia pugettensis replaced C. capitata as the numerically dominant organism at the site closest to the CSO. This switch in dominance may at least be partially explained by a change in habitat characteristics and food quality in the area. During high flows, heavy deposition of organic debris favors subsurface deposit feeding organisms such as C. capitata. When flows are less frequent, deposition and disturbance are correspondingly reduced. N. pugettensis is a surface dwelling organism that will swarm to areas containing large amounts or organic debris (Schmitt 1965). Because disturbance and deposition are reduced in August, this species is not buried or washed away from the area. The decline in the C. capitata population may be due to natural die off, competitive inferiority to N. pugettensis, or changed environmental conditions near the CSO.

Classification analysis results showed that depth produced the primary differences in community composition among sites and that effluent flows were of secondary importance. If the transect furthest from the CSO can be regarded as a control, stratification by depth in species composition by natural phenomena exists in spring but not in summer. Evidence from the classification and the proportional analyses suggests that the effluent has its primary effect at the 9 m sites on transects 2 and 5. Sites located between these latter two are somewhat less affected. Sediment characteristics indicate that effluent material is being deposited in greater quantities on transect 2. It has been shown that a lens of freshwater exists near the CSO during overflows and that the water in the lens is turbid with effluent particulate (METRO, unpublished data). Northward flowing currents may transport this material along the shore until mixing (possibly influenced by the small cove near transects 2 and 3) of the waters causes the material to be deposited on the bottom.

The patterns in the parameters relative to CSO disturbance follow the models summarized by Stirn et al. (1975) for subtidal infaunal communities. Recent work has indicated that these patterns are largely explained by the relative ability of the members of the local species pool to feed successfully in the disturbed area (Word 1978). Burrowing deposit feeding species species can withstand the constant rain of organic matter, whereas surface deposit and filter feeding organisms are not well suited for this type of habitat. Word (1978) has developed an index that utilizes the relative abundances of selected infaunal taxa of various feeding types to illustrate the degree of organic enrichment of benthic areas near sewage outfalls. The index is highly sensitive to changes in organic content of the sediments and shows that burrowing deposit feeding species predominate in areas of sewage deposition. The patterns in feeding types demonstrated by Word in southern California agree well with those seen by us near the Denny Way CSO.

We conclude that shallow subtidal infaunal communities subject to substantial variations in wastewater flows will respond to a measurable degree in concordance with these variations. The areas closest to the wastewater source will exhibit the greatest fluctuations, and the magnitude of the temporal changes will decrease with decreasing degree of influence (eg. distance). The resiliency of the benthic community we studied was evidenced by a slight recovery of the severely impacted areas during the period of infrequent discharges. We suggest that studies evaluating the impact of human disturbance on the marine environment, in regions where significant seasonal variations in physical-chemical parameters occur, should be conducted during several seasons. Results will vary by season in these areas and conclusions drawn on one season of sampling may be misleading regarding the overall impact of the pollution source on the environment.

ACKNOWLEDGEMENTS

We gratefully acknowledge G. Farris and R. Tomlinson of METRO for their assistance and technical support. We thank J. Packer and K. Li for their help with the field and lab work. C. Staude kindly verified the amphipod identifications. The typing and editorial assistance of J. Norton is greatly appreciated. This work was supported by a grant from the U.S. Environmental Protection Agency through the Municipality of Metropolitan Seattle (METRO).

LITERATURE CITED

Armstrong, J.W., R.M. Thom, K.K. Chew, B. Arpke, R. Bohn, J. Glock, R. Hieronymus, E. Hurlburt, K. Johnson, R. Mayer, B. Stevens, S. Tettlebach, and P. Waterstrat. 1978. The impact of the Denny Way combined sewer overflow on the adjacent flora and fauna in Elliott Bay, Puget Sound, Washington. A report prepared in cooperation with the Municipality of Metropolitan Seattle, 102p. (unpublished).

Armstrong, J.W., R.M. Thom, and K.K. Chew. 1980. Impact of a combined sewer overflow on the abundance, distribution and community structure of subtidal benthos. Marine Environmental Res. (in press).

Boesch, D.F. 1977. Application of numerical classification in ecological investigations of water pollution. U.S. Environmental Protection Agency, EPA-600/3-77-003, 115p.

Clifford, H.T. and W. Stevenson. 1975. An introduction to numerical classification. Academic Press, New York, 229p.

Hurlburt, S.H. 1971. The non-concept of species diversity: a critique and alternative parameters. *Ecol*. 54: 577-586.

Jumars, P.A. and K. Fauchald. 1976. Between community contrasts in successful polychaete feeding strategies. *Belle W. Baruch Libr. Mar. Sci.* 6: 1-20.

Littler, M.M. and S.N. Murray. 1975. Impact of sewage on the distribution, abundance and community structure of rocky intertidal macro-organisms. *Mar. Biol.* 30: 277-291.

Morgans, J.F.C. 1956. Notes on the analysis of shallow-water soft substrate. *J. An. Ecol.* 25: 367-387.

Reish, D.J. 1959. An ecological study of pollution in Los Angeles-Long Beach horbors, California. *Occ. Pap. Allan Hancock Foundation* 22: 1-119.

Schmitt, W.L. 1965. *Crustaceans*. The University of Michigan Press, Ann Arbor, 204p.

Smith, R.W. and C.S. Green. 1976. Biological communities near submarine outfalls. *J. Wat. Pollut. Control Fed.* 48: 1894-1912.

Snee, R.D. 1974. Graphical display of two-way contingency tables. *Amer. Statistician* 28: 9-12.

Sokal, R.R. and C.D. Michener. 1958. A statistical method for evaluating systematic relationships. *Univ. Kans. Sci. Bull.* 38:1409-1438.

Stevenson, W., R.W. Smith, C.S. Green, T.S. Sarason, and D.A. Hotchkiss. 1975. Soft bottom benthos from an area of heavy waste discharge: hierarchial classification of data. Southern California Coastal Water Research Project, El Segundo, California TM 226, 38p.

Stirn, J., A. Avcin, I. Kerzan, B.M. Marcotte, N. Meith-Avcin, B. Vriser, and S. Vukovic. 1975. Selected biological methods for the assessment of marine pollution. Pages 307-327 *In* E.A. Pearson and E.D. Frangipane, eds., Progress in water technology, marine pollution and marine waste disposal (Suppl.) Proceedings of the 2nd Inernational Congress, San Remo. Permagon Press, New York, 487p.

Word, J.Q. 1978. The infaunal trophic index. Pages 19-39 *In* W. Bascom, ed., Coastal Water Research Project Annual Report 1978. Southern California Coastal Water Research Project, El Segundo, California, 253p.

PRODUCTIVITY RESPONSES OF LAKE EOLA TO URBAN RUNOFF

by

Harvey H. Harper, III, Yousef A. Yousef, and Martin P. Wanielista
Civil Engineering and Environmental Sciences
University of Central Florida
Orlando, Florida 32816

ABSTRACT

Lake Eola is a land-locked lake located in downtown Orlando, Florida. It is a focal point for the city and is visited by many tourists for its aesthetic appeal. No swimming, motor boating, or fishing is allowed. The lake was drained down exposing 40% of the bottom and approximately 4 feet of mud was removed in 1972-1973. However, input nutrients, organics and other compounds from street stormwater runoff have not been effectively managed. Five years later, excess fish and algae have to be removed periodically and the water quality of Lake Eola is again questioned.

A research project is sponsored by the U.S. Environmental Protection Agency, the Florida State Department of Environment Regulation and the City of Orlando to determine the lake impacts of pollutants in stormwater runoff optimal combinations of stormwater management methods.

Periodical water samples are being collected from the lake for various stormwater events. Changes in water quality parameters with time through each storm event are documented. Parameters analyzed include alkalinity, hardness, solids, BOD_5, TOC, nitrogen, phosphorus, and heavy metals for particulate and dissolved fractions. Loading rates from nutrients and heavy metals release to Lake Eola due to stormwater runoff are analyzed, and lake impacts are evaluated.

Algal bioassay studies are performed to investigate stormwater impacts on algal productivity. Periodical water samples are being collected from various locations in the lake, mixed and filtered for limiting nutrient studies using various concentrations of N, P, and Fe. Unialgal species of Selenastrum, Cholorella and indigenous species are used and changes in chlorophyll "a" and biomass are measured. Initial results indicate that phosphorus or nitrogen can be limiting at sometimes of the year. Maximum standing crop seem to occur at N:P ratio of 15-20:1.

Also, similar bioassays are performed on a mixture of stormwater, coagu-

lated stormwater and lake water at different ratios. Higher concentrations of stormwater would inhibit algal productivity and smaller ratios would significantly increase productivity. Additionally, no increase in productivity is observed when coagulated runoff water is used. These experiments can be used as useful tools to facilitate stormwater management decisions.

INTRODUCTION

Lake Eola is a small land-locked lake located in downtown Orlando, Florida. The lake receives direct stormwater runoff from a watershed of approximately 78.2 acres of commercial and 57.8 acres of residential areas surrounding the lake. There are currently eleven active street drains which drain stormwater directly into the lake (Figure 1). The level of the lake is maintained between 87.0 and 88.5 feet above sea level by two drainage wells which drain excess water into the underlying artesian aquifer. The natural shoreline of the lake has been replaced with a stone wall to prevent flooding of the adjacent parkland, and numerous small patches of rooted emergent macrophytes exist along this wall. Howrver, no rooted submergent plants of any kind have been noted in the lake. Physical characteristics of Lake Eola are listed in Table 1.

Floating masses of dead algae and fish and their accompanying odor are a common occurrence in Lake Eola. With the exception of areas near the shoreline, the bottom of the lake is covered with an accumulation of loose flocculant partially decomposed organic matter which is easily disturbed. The loose nature of this material makes it difficult for rooted submergent plants to exist and with the exception of a very small area near the shoreline, no rooted plants of any kind were seen in Lake Eola. In areas near the center of the lake this organic matter, subjected to long periods of anoxis and reducing conditions, has formed into sapropen complete with the characteristic hydrogen sulfide smell.

A restoration project of Lake Eola was undertaken in 1972. At that time the lake was partially drained and approximately 40% of the bottom was cleaned of muck and silt and covered with relatively clean builders' sand. Existing stormwater drains were also extended into the lake approximately at the 8-10 foot water level. The lake was then refilled with water from the drainage wells. However, no efforts were undertaken at that time to manage or treat the stormwater entering the lake, and now the water quality of Lake Eola is again questioned. Buildup of flocculant sediment matter is increasing rapidly. Large masses of algae can be seen floating along the shoreline, and fish and duck kills have been reported periodically during spring and summer months following heavy rain events. Also, *Salmonella*, *Shigella* and *Clostridium botulinum* have been isolated from the water and shoreline sediments in the lake.

The lake is a focal point and tourist attraction in the heart of downtown Orlando, Orange County, FLorida. Therefore, a research project was initiated during 1978 and supported by the US EPA, Florida Department of Environmental Regulation, City of Orlando, and the Engineering and Industrial Experiment

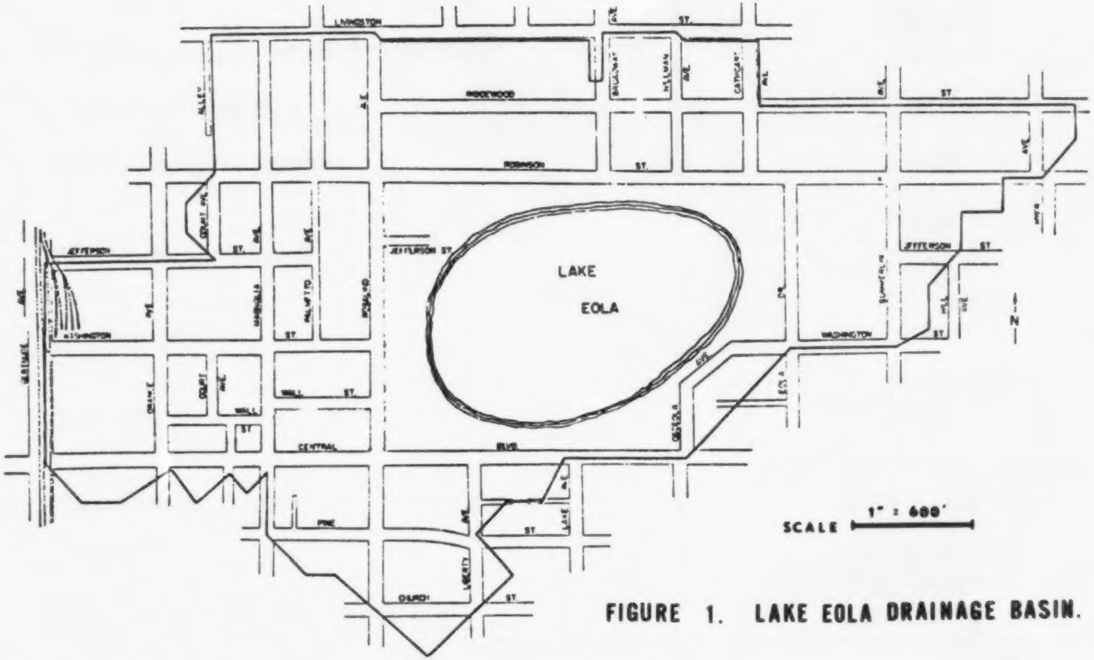


FIGURE 1. LAKE EOLA DRAINAGE BASIN.

TABLE 1

PHYSICAL CHARACTERISTICS FOR LAKE EOLA, FLORIDA

Parameter	Quantity
Approximate Surface Area	109,270 m^2 0.92 hectares (27.00 acres)
Approximate Volume	3.30 x 10^5 m^3 87.3 x 10^6 gallons
Mean Depth	3.02 m (9.92 ft)
Maximum Depth	6.8 m (22.3 ft)
Length of Shoreline	1417 m (4650 ft)
Shoreline Development	1.21
Volume Development	1.72
Average Height Above Sea Level	26.8 m (88 ft)

Station at the University of Central Florida to develop:

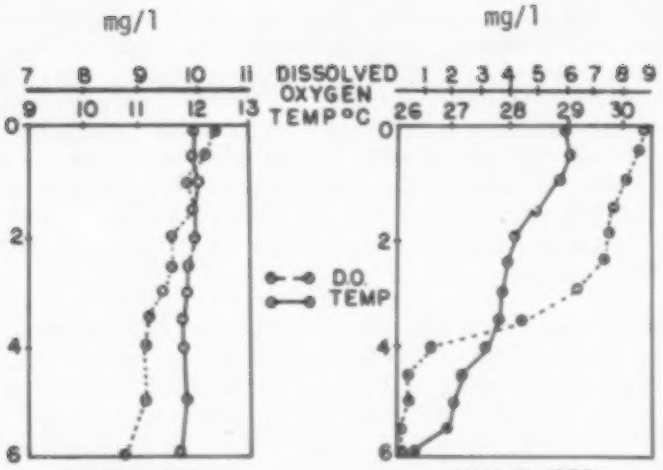
1. The nature and extent of pollutional loads from stormwater runoff to Lake Eola
2. Impact of these loadings on the lake ecology
3. Management techniques to minimize stormwater effects
4. A proposed plan for the restoration of Lake Eola

EXPERIMENTAL METHODS

Water samples were collected from stormwater drains during stormwater events at selected time intervals to determine pollutional loads released to the lake. Also, Lake Eola was divided into six representative locations for field measurements and sampling for water quality analysis and bioassay experimentation. Water quality analyses were performed at each station on a monthly basis for a period of one year beginning in July, 1978. All samples were collected from a depth of one meter using a 2.0 liter Kemmerer water sampler, stored in one gallon polyethylene containers which were completely filled to eliminate gas exchange and placed on ice in the dark for return to the laboratory. Dissolved oxygen and temperature profiles were recorded monthly at 0.5 m intervals and at the bottom using a YSI Model 54A Dissolved Oxygen Meter equipped with a remote sensing probe. Seechi disk depth was also determined at each sampling station.

Water samples were collected and returned to the Environmental Engineering Laboratory at the University of Central Florida for water quality analysis and for use in bioassay experiments. The following determinations were performed on each sample collected: pH, turbidity, organic carbon, inorgainc carbon, nitrate nitrogen, orthophosphorus and chlorophyll "a" along with an analysis of heavy metals which included: zinc, lead, chromium, nickel, copper, aluminum, iron, cadmium and arsenic. Total Kjeldahl nitrogen and BOD_5 were also determined on selected samples. Measurements of pH, turbidity, and chlorophyll "a" were performed within 4 hours of collection. Other analyses were conducted within the time specified by EPA in Methods for Chemical Analysis of Water and Wastes (1976). Chlorophyll "a" concentrations were determined from a calibration curve using a Turner Model 111 Filter Fluorometer. The calibration curve was prepared by calculateing chlorophyll "a" concentrations in water samples from Lake Eola using the trichromatic spectrometric acetone extraction method as described in Standard Methods for the Examination fo Water and Wastewater, 14th Edition and comparing these values to relative fluorescence values.

Heavy metal analyses were performed on concentrated samples using a Spectrometrics Incorporated Spectrospan III Plasma Emission Spectrophotometer. Samples were concentrated by adding 2.0 ml of concentrated HNO_3 to 100 ml of sample in a 250 ml Erylenmeyer flask and heating at 95°C until a volume of approximately 10 ml was achieved. The sample was then brought up to 20.0 ml with glass distilled water and stored in a covered polypropelene container for measurement. All glassware used in metal determinations was acid-washed before each use with a 1:1 solution of hot hydrochloric acid followed by 5 rinses in glass distilled water.



FEB 10, 1979

JULY 10, 1979

FIGURE 2. VARIATION IN TEMPERATURE AND DISSOLVED OXYGEN WITH DEPTH IN LAKE EOLA IN THE WINTER AND THE SUMMER.

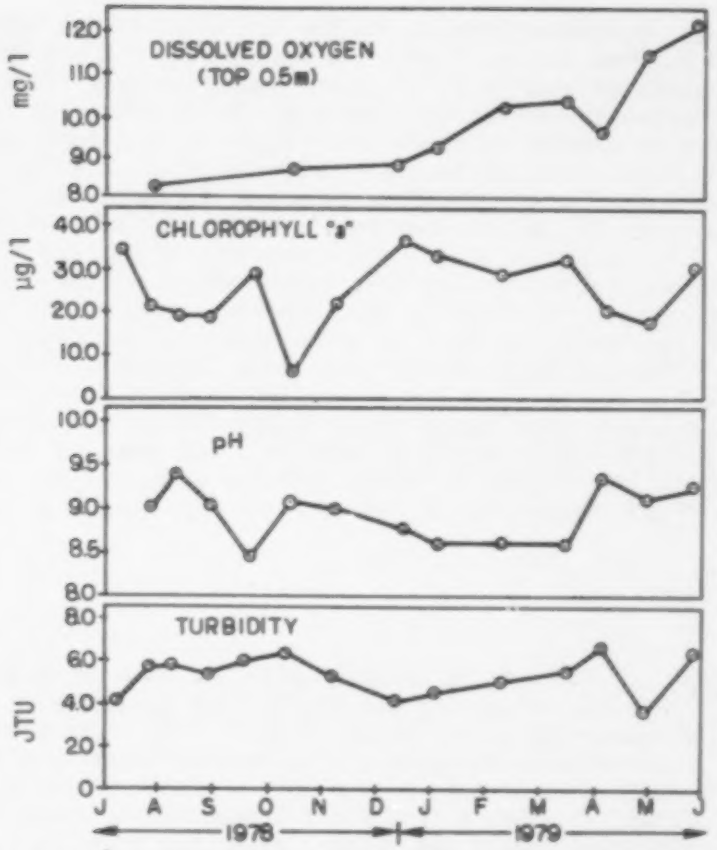


FIGURE 3. AVERAGE PHYSICAL CHEMICAL PARAMETERS OF WATER SAMPLES COLLECTED FROM THE TOP 0.5m IN LAKE EOLA.

FIELD ANALYSIS

Temperature profiles in Lake Eola indicate little variation between the stations recorded on a specific date and variations in temperature between stations were generally limited to $1^{o}C$ or less. The highest temperature recorded during the test period was $30.9^{o}C$, occurring during August. Lowest lake water temperature was measured during February with a value of $11.1^{o}C$. Seechi disk measurements at stations fluctuated slightly during the test period with recorded values between 0.8-1.3 meters, the lowest values occurring during the period of September to December.

Dissolved oxygen concentrations in the top 0.5 meter of the water column fluctuated between 7.5 and 12.6 mg/l over the test period. Concentrations of dissolved oxygen, although usually at or above saturation near the surface, drop periodically during the spring and summer months to 1 mg/l or less at depths of 4.0 meters or greater as shown from Figure 2. Although measurements of dissolved oxygen have indicated that the deeper areas in the lake can occasionally become oxygenated by wind action, it seems reasonable to assume that areas in Lake Eola below 4-5 meters deep remain anoxic during much of the spring and summer. Besides releasing large quantities of H_2S and CO_2 from bottom sediments, these conditions can also cause release of considerable quantities of phosphorus. This release from sediment material was measured in Lake Eola to be as high as 250 mg-P of orthophosphorus per meter square after approximately two months of anoxic conditions (Marshall, 1979). If this release is conservatively considered to be limited to areas greater than 4.5 meters deep, the area under this depth would be approximately 30,000 square meters and the expected release may be as high as 7,500 grams of orthophosphorus. A release of this magnitude, if mixed into the water column, would increase average in situ orthophosphorus concentration by as much as 0.023 mg/l-P.

LABORATORY ANALYSES

Lake Eola water was found to be somewhat alkaline with pH values ranging from 8.3 to 9.5 (Figure 3), presumably due to the high rate of algal production. At the average pH value in Lake Eola of 8.86, approximately 96% of the inorganic carbon present would exist in the bicarbonate state. The remaining 3% would be in carbonate form with only a minute percentage of free CO_2 in Lake Eola. Thus, it appears that algal production is depleting this source almost as fast as it enters the water. After the free CO_2 is utilized, those photosynthetic organisms which are capable of utilizing bicarbonate ions also begin to do so. It is not unreasonable to assume that the composition of algal communities in Lake Eola is determined and regulated not only be seasonal variations but also by the type of inorganic compounds present. PH values were typically lower in fall and winter months when algal production would be expected to decrease and higher in the spring and summer when production is at a maximum. Turbidity increases during spring and summer (Figure 3) are also indicative of this increased production.

Nutrient data for Lake Eola, in many cases, does not follow typical predicted cyclic patterns, which is to be expected considering the irregularity in both quantity and quality of the pollution source (Figure 4). Concentrations of orthophosphorus fluctuated between 0.01 and 0.04 mg/l during the test period. Although concentrations would normally be expected to reach minimum values during the highly productive spring and summer months, peaks in phosphorus were found in August and May, presumably due to stormwater additions and/or phosphorus release from bottom sediments. Organic carbon was lowest in the winter months as expected due to decreases in algal production and stormwater inputs with a corresponding increase during the spring and summer. However, a very large increase in organic carbon was measured during October and November and was presumed to be due to mixing of sediment material caused by circulation increases typical of the fall season. Inorganic carbon appeared to be lower during the spring and summer as algae utilized this substrate as a food source with slight increases during the nonproductive winter months. A two-fold increase in concentration was recorded in April 1979 presumably due to stormwater inputs. Although the data on nitrate nitrogen is limited, it appears that nitrate concentrations experienced decreases during spring and summer as algae utilization increased. Concentrations of chlorophyll "a" were extremely varied over the test period. Values measured at individual stations ranged between 6.2 μg/l during October to 41.4 μg/l in July. Although the data is very limited, concentrations of BOD_5 seemed to be very low. Average BOD_5 at individual stations ranged from 3.0 to 4.5 mg/l.

Total metal concentrations measured in composite water samples collected in Lake Eola are listed in Table 2. No apparent trend was observed from the data presented and the fluctuations seem to remain within a narrow range and oscillate around an average value.

BIOASSAY EXPERIMENTATION

Bioassay experiments were conducted to evaluate algal responses to nutrient changes in Lake Eola water and to study the impact of untreated and coagulated stormwater runoff on algal cell production. In this research, four different types of sample preparations were used: raw unfiltered lake water, water filtered through a 0.45 micron Millipore filter (F), water which was filtered then autoclaved (F/A), and water which was autoclaved then filtered (A/F). In addition to indigenous algal species from Lake Eola, two different laboratory cultured test species were also utilized, *Chlorella pyrenodosa* and *Selenastrum capricornutum*. Both of these species are solitary, non-motile green algae which posses a wide tolerance towards environmental conditions and occur in waters of diversified composition (EPA 1978). A summary of experimental conditions for bioassay studies conducted during this research is listed in Table 3.

The validity of results obtained from a bioassay experiment hinges upon the ability of the experiment to determine the maximum growth response and growth limiting nutrients of a test water. Since these responses are an integration of the combined effects of ion solubility and ion availability to the test organism, any factor which alters these conditions, such as sample treat-

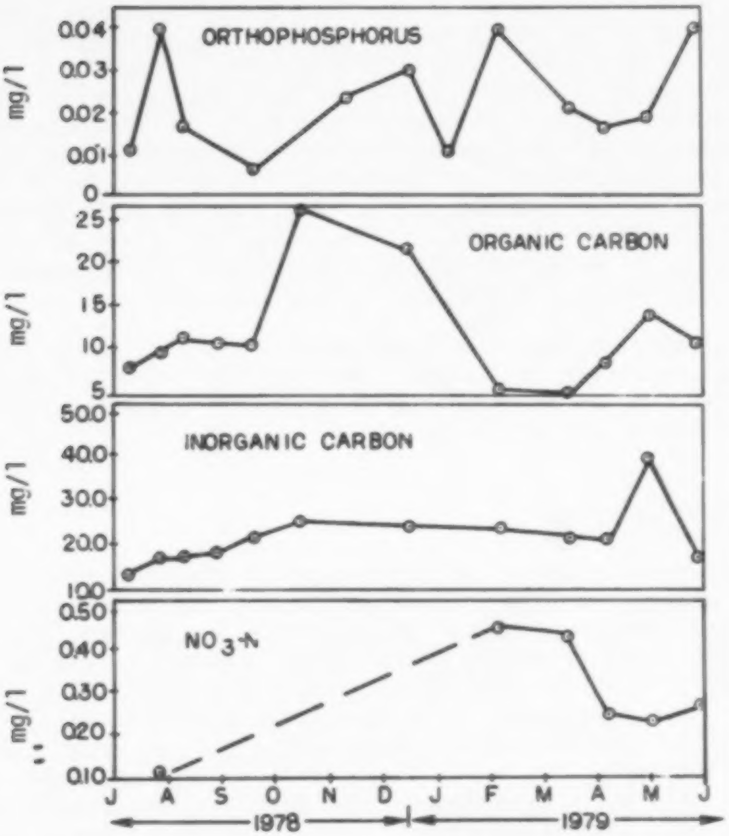


FIGURE 4. AVERAGE PHYSICAL CHEMICAL PARAMETERS OF WATER SAMPLES COLLECTED FROM THE TOP 0.5m IN LAKE EOLA

TABLE 2

TOTAL METAL CONCENTRATIONS IN COMPOSITE WATER SAMPLES
COLLECTED FROM THE TOP 0.5 m IN LAKE EOLA

Date	Metal Concentration (mg/1)									
	Al	Fe	Pb	Cr	Zn	Cd	As	Ni	Cu	Ca
2/10/79	0.173	0.138	0.084	0.025	0.039	0.002	0.029	0.008	0.033	----
3/13/79	0.134	0.141	0.077	0.029	0.054	0.002	0.045	0.009	0.060	----
4/06/79	0.384	0.136	0.063	0.035	0.025	0.000	0.116	0.017	0.053	----
4/30/79	0.235	0.121	0.069	0.020	0.015	0.000	0.100	0.005	0.041	----
6/20/79	0.236	0.045	0.036	0.030	0.020	0.000	0.000	0.006	0.046	21.3
7/10/79	0.226	0.197	0.083	0.032	0.055	0.001	-----	0.005	0.057	25.1
Average	0.231	0.130	0.069	0.029	0.035	0.001	0.058	0.008	0.058	23.2

TABLE 3

SUMMARY OF EXPERIMENTAL CONDITIONS FOR BIOASSAY STUDIES

Date	Lake Water Treatment	Test Organism	Runoff %	Nutrients Added mg/1			EDTA Added mg/1
				N	P	Fe	
8/10/78	Composite F	*Chlorella*	0	1-5	0.1-1.0	0	0
10/16/78	Composite F/A	*Chlorella*	0-75	0	0	0	0
11/08/78	Composite F	*Chlorella*	0-100	0	0	0	0
12/14/78	Composite F	*Chlorella*	0-100 (dry weather)	0	0	0	0
1/14/79	Composite F/A	*Selenastrum*	0	1-5	0.05-0.5	0.05	0
2/21/79	Composite A/F	*Selenastrum*	0	1-5	0.05-0.5	0.05	0
	Composite F/A		0	1-5	0.05-0.5	0.05	1
3/18/79	Composite A/F	*Selenastrum*	0	1-10	0.5	0	0
	Composite F/A		0	1-10	0.5	0	0
4/30/79	Composite	Indigenous	0-50 (untreated)		0	0	0
	Composite	Indigenous	0-50 (alum coagulated)	0	0	0	0
7/17/79	Composite	Indigenous	0-50	0	0	0	0
	Composite	Indigenous	0-50 (Fe Coagulated)	0	0	0	0

ment, may bias bioassay results. Chemical analyses conducted during this research indicate that autoclaving, filtration followed by autoclaving, and autoclaving followed by filtration all result in significant decreases in concentrations of organic carbon, inorganic carbon, and orthophosphorus. However, nitrate nitrogen concentrations show substantial increases due to these treatments (Table 4). Similar results have been reported by Filip and Middlebrooks (1973) in tests on mesotrophic pond water. They suggested that oxidation attributable to the physical disturbance of the sample as it passes through the filter probably accounts for higher nitrate levels in treated samples. It seems reasonable that increases in temperature and pressure during autoclaving may also be responsible for chemical changes of various parameters in water samples. Oxidation of organic carbon, nitrogen and phosphorus may occur altering the composition and quantity of these nutrients while at the same time, slightly increasing the pH. Autoclaving of the sample either before or after filtration, besides substantially reducing concentrations of organic carbon, inorganic carbon, and orhtophosphorus can also cause reductions in concentrations of calcium, phosphorus, iron, and lead by precipitation during autoclaving (Filip and Middlebrooks, 1975). These precipitates may be resistant to resolubilization under bioassay conditions (Environmental Protection Agency, 1978). Oxidation of organic wastes may also occur rendering these compounds less toxic. It appears that autoclaving of a sample tends to produce conditions which may enhance algal production by increasing the availability of some nutrients and at the same time decreasing concentrations of organic compounds and certain metals. The maximum yields obtained in this research using test waters which were autoclaved then filtered were much larger than any other type of treatment.

From the results obtained during this research, it appears that autoclaving of a sample may not be a suitable treatment technique for use in studies where the effects of complex organic compounds are to be determined. In these cases, filtration should be considered. If, on the other hand, a relatively clean oligotrophic water is to be assayed and it is desired to determine the amount of algal biomass that can be grown from all nutrients in the water, including thos contained in filterable organisms and other particulate matter, then autoclaving of the sample seems to be the proper technique. As a general rule, samples which are autoclaved will produce a much larger standing crop than filtered samples.

ALGAL RESPONSES TO NUTRIENT ADDITION

A summary of nutrient addition bioassay experiments using Lake Eola water is listed in Table 6. As seen in this data, both nitrogen and phosphorus were able to stimulate algal production in Lake Eola on certain dates. Although the data is limited, it appears that stimulation of a particular water by nitrogen or phosphorus may be related to the background phosphorus concentration at the beginning of the bioassay. Whenever the orthophosphorus concentration before nutrient additions was approximately 0.02 mg/l or less, algal production was stimulated by a phosphorus addition. Above this value, nitrogen was shown to stimulate growth. Typical response curves of Selenastrum, to various nutrients added to Lake Eola water are shown in Figure 5.

TABLE 4

PERCENT CHANGE IN WATER QUALITY PARAMETERS DUE TO SAMPLE TREATMENT PRIOR TO BIOASSAY USE

Sample Treatment	# of Samples	Percent Change Due to Treatment				
		pH	Organic Carbon	Inorganic Carbon	NO_3-N	O.P.
Filtered	2	-1.5 (-2,-1)	-12.5 (-20,-5)	- 1.5 (0,-3)	+ 7.5 (0,15)	-58.0 (-58)
Autoclaved	2	+1.0 (-1,+3)	-53.0 (-96,-10)	-16.0 (-37,+5)	+17.0 (+15,+19)	-21.5 (-27,-16)
F/A	3	+3.3 (+6,+2)	-40.0 (-100,-6)	-38.0 (-50,-22)	+48.0 (+25,+65)	-39.0 (-94,-3)
A/F	4	+1.0 (-3,+4)	-36.5 (-98,-5)	-35.3 (-53,-2)	+25.0 (+4,+46)	-18.0 (-33,-3)

* Range of values shown in ().

TABLE 5

SUMMARY OF BIOASSAY RESULTS OBTAINED IN CONTROL FLASKS CONTAINING LAKE EOLA WATER WITH NO NUTRIENT ADDITIONS

Date of Sample Collection	Sample Treatment	Maximum Cell Yield Obtained in Control Flasks (mg/1)	Test Organism	*In Situ* Conc.		Ratio of CHYL "a"/Max Yield	Ratio of Cell Mass Experimental /*In Situ*
				CHYL "a" (µg/1)	Cell Mass mg/1		
8/10/78	F	0.75	*Chlorella*	19.4	0.66	25.9	1.1
11/08/78	F	3.39	*Chlorella*	23.3	0.79	6.9	4.3
10/16/78	F/A	0.47	*Chlorella*	6.7	0.23	14.3	2.0
1/14/79	F/A	0.90	*Selenastrum*	33.6	1.14	37.7	0.8
3/18/79	F/A	2.10	*Selenastrum*	32.4	1.09	15.4	1.9
2/21/79	A/F	10.10	*Selenastrum*	28.2	0.96	2.8	10.5
4/30/79	none	0.83	Indigenous	17.5	0.59	21.1	1.4
7/17/79	none	0.91	Indigenous	17.9	0.61	19.7	1.5

A close examination of the data indicates that the N:P ratio may, under certain conditions, be more influential in regulating algal production in Lake Eola than the actual concentration of nitrogen or phosphorus alone. The fact that N:P ratios can play an important part in determining the potential biomass in a system is clearly demonstrated in Figure 6. In these experiments an optimum weight ratio of total soluble inorganic nitrogen (TSIN) to orthophosphorus was found to be between 7.4 and 21.4. Chiaudani and Vighi (1974) report optimum weight ratios between 4.5 and 9 in studies on Italian lakes. It seems reasonable to assume that when a nutrient is present in a minimal quantity that a point will be reached in terms of the concentration of a particular nutrient at which the availability of the nutrient will become so low that further growth will be limited by ion availability rather than by its relationship to other ions. It appears then that a threshold level of phosphorus exists below which the growth of an organism is regulated solely by the concentration of phosphorus present, assuming all other essential nutrients are also present. Above this threshold value, phosphorus is in relative excess and the organism is no longer limited by ion availability alone. As concentrations of ions increase, organisms are virtually surrounded by an abundance of nutrients. Since some molecules have greater affinities for binding sites on the cell membrane than others, these binding sites may become blocked by certain compounds, limiting further nutrient uptake. In an environment with an excess of nutrients, cell growth will be optimum at a nutrient ratio where nutrients needed for growth can be readily taken up by the cell in approximate quantities necessary for growth. As seen in Table 6, an orthophosphorus concentration of 0.060 mg/l was not sufficient to produce excess ion availability. However, when the concentration is increased to 0.110 mg/l as was the case in the experiment using lake water collected on 3/18/79, optimum growth occurred at a weight ratio of 6.4:1, which is within the range previously reported. It appears then that the threshold concentration within Lake Eola above which sufficient phosphorus ions are available so that growth may be limited by the relative abundance of these ions rather than their actual concentrations is between 0.060 and 0.110 mg/l PO_4-P. In other words, when the concentration of orthophosphorus in Lake Eola is approximately 0.10 mg/l or less, algal production is regulated by the concentration of orthophosphorus alone. Above this concentration, it is assumed that an excess of phosphorus is available and algal growth is regulated by the relative availability of nitrogen or phosphorus. However, since concentrations of orthophosphorus recorded in Lake Eola during this research were 0.04 mg/l or less, it can be concluded that, except during periods of heavy P loadings by stormwater inputs or by long periods of anoxia, algal production in Lake Eola is regulated by the concentrations of added phosphorus alone.

STORMWATER IMPACT

Average water quality characteristics for runoff water from Lake Eola drainage basin for eight storm events are presented in Table 7. Water quality parameters seem to fluctuate over a range of more than ten fold in some parameters such as TOC and NO_3^-. Also, more than half of the nutrients released to the lake in stormwater runoff appear to be in solution. Calculations of the mass loadings from these storm events are presented in Table 8. Mass

TABLE 6

SUMMARY OF NUTRIENT ADDITION BIOASSAY EXPERIMENTS USING LAKE EOLA WATER

Sample Characteristics				Maximum Experimental Yield (mg/l)	Predicted Maximum Yield (mg/l)	Nutrient Addition Producing Maximum (Yield)	N:P Ratio at Maximum (Yield)	Test Organism
Date Collected	Treatment	PO_4-P (mg/l)	NO_3-N (mg/l)					
8/10/78	Filtered	0.017	-----	1.63	10.97	1.0 mg/l N	-------	*Chlorella*
1/14/79	F/A	0.011	-----	2.38	39.35	0.05 mg/l P	-------	*Selenastrum*
2/21/79	A/F	0.060	1.25	21.00	38.70	3.0 mg/l N	71:1	*Selenastrum*
2/21/79	A/F (EDTA Added)	0.060	1.25	39.50	38.70	5.0 mg/l N	104:1	*Selenastrum*
3/18/79	F/A	0.010	0.58	4.70	65.15	0.1 mg/l P	6.4:1	*Selenastrum*

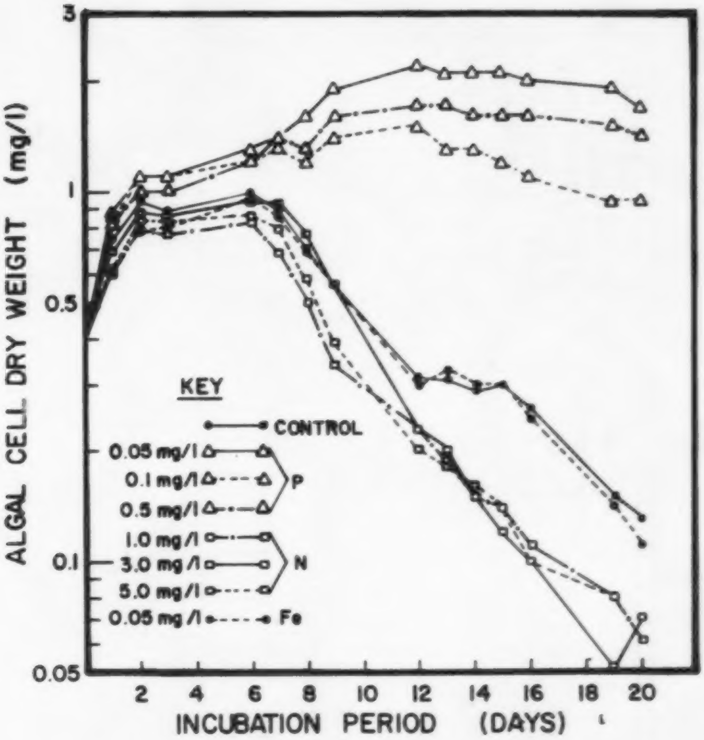


FIGURE 5. RESPONSES OF *SELENASTRUM* TO VARIOUS NUTRIENTS ADDED TO LAKE EOLA WATER SAMPLES COLLECTED 1-14-79 (LAKE WATER FILTERED THEN AUTOCLAVED).

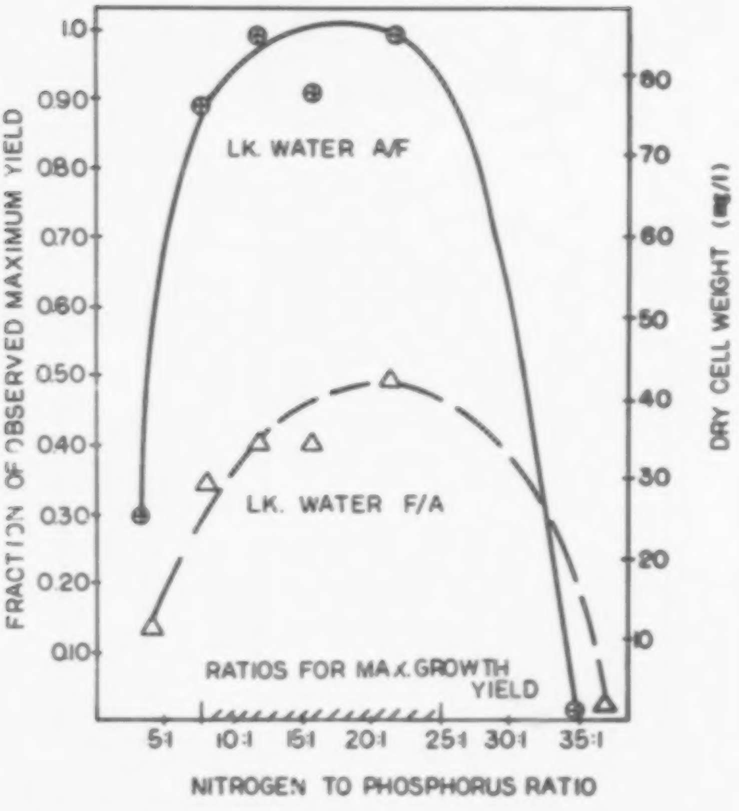


FIGURE 6. CHANGES IN MAXIMUM GROWTH YIELD IN A SELENASTRUM CULTURE INCUBATED IN LAKE EOLA WATER DUE TO CHANGES IN NITROGEN TO PHOSPHORUS RATIOS. (LAKE WATER COLLECTED 3-18-79).

TABLE 7

AVERAGE WATER QUALITY CHARACTERISTICS OF URBAN STORMWATER
COLLECTED FROM LAKE EOLA DRAINAGE BASIN

Date	Rainfall (inch)	pH	Spec. Cond. µmho/cm	Alk.	Turb JTU	SS mg/1	BOD_5 mg/1	TOC mg/1		Nitrogen mg/1-N		Ortho-Phosphorus mg/1-P
								Total	Diss.	TKN	NO_3^-	Dissolved
7/11/78	0.18	6.9	---	34.8	26.9	142	10.0	33.4	26.6	4.35	1.33	0.44
7/14/78	0.11	7.8	209	59.6	18.9	77	22.0	38.2	30.1	2.14	0.55	0.25
7/25/78	3.20	7.6	66	134	31	129	5.7	21.8	10.2	0.65	0.42	0.10
8/03/78	0.04	7.6	71	25	24	88	9.0	30.6	13.1	--	0.62	0.15
8/10/78	0.43	7.3	38	15	14	74	4.6	18.9	10.1	0.65	0.29	0.13
8/15/78	0.18	7.5	193	69	44	134	16.3	178	137	1.04	3.20	0.67
10/12/78	0.10	7.4	256	65	65	275	16.4	284	248	3.30	0.39	0.28
11/07/78	0.04	8.3	337	95	28	27	18.3	218	206	3.29	8.90	0.53

TABLE 8

MASS LOADINGS FROM LAKE EOLA BASIN

Date	Mass Loading, kg/ha-cm			
	SS	BOD_5	TOC	Soluble Ortho-Phosphorus
7/11/78	18.5	2.6	4.1	0.06
7/14/78	4.4	0.6	6.0	0.02
7/25/78	8.9	0.3	1.0	0.02
8/03/78	9.3	0.9	3.6	0.02
8/10/78	4.5	0.3	1.6	0.03
9/15/78	5.9	2.0	20.0	0.02
10/12/78	13.0	1.0	18.0	0.02
11/07/78	1.8	1.1	13.4	0.03
Average	8.3	1.1	8.5	0.03
Calculated Loadings* (kg/yr)	58,432	7,744	59,840	211

*Assumes a drainage basin area of 55 ha and average rainfall of 128 cm/yr.

loading average 8.3, 1.1, 8.5, and 0.03 kg/ha/cm of rainfall for suspended solids, BOD_5, TOC, and orthophosphorus, respectively. Assuming the drainage basin area is 55 hectares and the average rainfall is 128 cm, the total soluble orthophosphorus released to the lake is 211 kilograms/yr. Similarly, loadings from suspended solids, BOD_5 and TOC will equal to 58, 432, 7,744, and 59,840 kilograms per year, respectively. If all the suspended solids are transported to the area of the lake deeper than 6.5 meters, a buildup of solids approximately 0.15 meter deep will accumulate per year in the deep areas of the lake. This material, in addition to rapidly decreasing available water depth in the lake, also creates a significant oxygen demand during its decomposition and it may be responsible for anoxic conditions during spring and summer months in the deep areas. Of the dates on which stormwater was collected the lowest concentrations of virtually every stormwater parameter seemed to occur during the months of July and August. These months are typically characterized by frequent and intense storm events. Since the dry period between storms is very short, the accumulation of transportable material in contributing areas of the watershed is minimized, and nutrient input into the lake per storm is reduced. However, since storm events occur almost daily during this period, the total input of nutrients over this rainy season will be very large although it will be stretched out over a period of several months. The frequent and intense nature of these storms serves to reduce concentrations of toxic heavy metals and organics while at the same time supplying the lake with a constant source of phosphorus, nitrogen, and carbon. This continuous supply of nutrients with reduced concentrations of toxic elements combined with warmer water temperatures during these summer months provides enriched conditions for algal growth, and one would expect the most rapid rate of algal growth to occur during this time. However, because dilution of algal populations by stormwater and the constant removal of the upper layers on the lake via the drainage well are constantly lowering these populations, the highly enriched growth which is occurring during this period is not reflected by in situ chlorophyll "a" concentrations.

In contrast to the enhanced algal growth conditions experienced during the summer rainy months due to stormwater events, runoff entering the lake after prolonged periods of drought may produce severe toxic effects on aquatic life in Lake Eola. Contaminants have been allowed to accumulate within the watershed over this period, and when a storm event occurs, the mass loading to the lake is many times larger than that experienced during a rainy period. This large influx of toxic and oxygen demanding wastes can be lethal to many forms of aquatic life. Evidence of such a phenomenon was recorded in March, 1979 when a rain event occurred after a 6 weeks dry spell. Concentrations of organic carbon as high as 400 mg/l were measured in stormwater runoff entering the lake during this event. Two days after this event, dissolved oxygen concentrations had been reduced from saturation near the surface to 4 mg/l at a depth of 1 meter and to virtually zero below 2 meters. Numerous large-mouth bass averaging 2-3 pounds were also found floating in the water, and large masses of dead filamantous algae had accumulated in thick mats over much of the lake's surface. The color of the water itself was changed from its characteristic blue-green tint to a grey-green appearance. Seechi disk depth was reduced to less than 0.5 meter. After approximately 5 days, conditions began to improve, and after 10 days physical conditions in the lake, as determined by dissolved oxygen profiles and Seechi disk measurements, had returned to near normal values for this time of year. However, it should not be inferred that the lake had returned to the same ecological condition as before the storm event. The damage caused to the lake system by increases in sediment buildup and loss of animal species are difficult to document.

A summary of stormwater addition bioassay experiments using Lake Eola water is listed in Table 9. As seen in this data, maximum yields were produced in virtually every case with a mixture of 25% stormwater runoff, resulting in increases of 87-731% over control flasks. If Figure 7 is considered a typical algal response to stormwater additions it can be seen that the addition of stormwater in any concentration, even 100% stormwater runoff, resulted in an increase in algal production over the control. However, when stormwater was added in concentrations of 25% or greater, the growth curve was typified by an initial die-off of algal populations, the magnitude of which being a function of runoff concentration. This die-off is due largely to the presence of toxic elements in the stormwater which is discussed, along with its implications, in a later section. If a typical ½ inch storm fell on the Lake Eola watershed the corresponding runoff, if completely mixed throughout the water column, would represent an addition of only 2% to the total lake volume. At this concentration of stormwater runoff the effect on algal production would be greatly minimized. However, the time necessary for the stormwater to mix throughout the entire lake, depending on weather conditions, is probably on the order of several days. It seems reasonable to assume then that stormwater concentrations of 25% or greater may exist near stormwater outfalls for at least 1 day and that the die-off predicted at this concentration would occur in these areas.

EFFECT OF COAGULATED RUNOFF ON ALGAL GROWTH

One of the techniques under consideration for the restoration of Lake Eola includes chemical coagulation of a portion of the runoff to remove phosphorus and heavy metals. Alum, $Al_2(SO_4)\cdot 18H_2O$, was selected as a coagulant for test purposes. An alum dosage of 240 mg/l at a final pH of 5.5 was determined through jar tests to provide optimum phosphorus removal. Coagulation under these conditions resulted in a reduction of 96% of orthophosphorus and 87% of nitrate nitrogen (Table 9). With the exception of aluminum and arsenic coagulation also reduced concentration of every heavy metal tested. As seen in Table 9 and Figure 7, coagulation of the stormwater resulted in substantially lower growth responses, as compared with uncoagulated test flasks, at every concentration of stormwater tested. The maximum percent increase in algal production over the control flasks was reduced from 731% using untreated stormwater to 8% in a coagulated sample.

An interesting result of these coagulation experiments is that the untreated sample required 11 days to reach its maximum yield while the coagulated sample obtained a maximum in only 6 days. Greene (1976) suggests that a growth lag in algal response is often experienced in waters containing toxic compounds. Certain algal forms are able to produce extracellular substances which can form chemical complexes with the growth inhibiting substance. The absence of this characteristic growth lag in the coagulated samples suggests that sufficient toxic elements had been removed by this process so that algal

TABLE 9

SUMMARY OF STORMWATER ADDITION EXPERIMENTS USING LAKE EOLA WATER

Sample Characteristics				Maximum Experimental Cell Yield (mg/1)	Stormwater Conc. Producing Maximum Yield	Test Organism	# of Days to Reach Maximum Yield	Percent Increase Over Control
Date Collected	Treatment	PO_4-P	NO_3-N					
10/16/78	F/A	0.480	--	3.27	25%	*Chlorella*	8	596%
11/08/78	F	0.650	--	6.33	25%	*Chlorella*	10	87%
4/28/79	None	0.291	6.90	3.72	25%	Indigenous Species	11	731%
4/28/79	Alum Coagulated	0/013	0.90	1.04	5%	Indigenous Species	6	8%

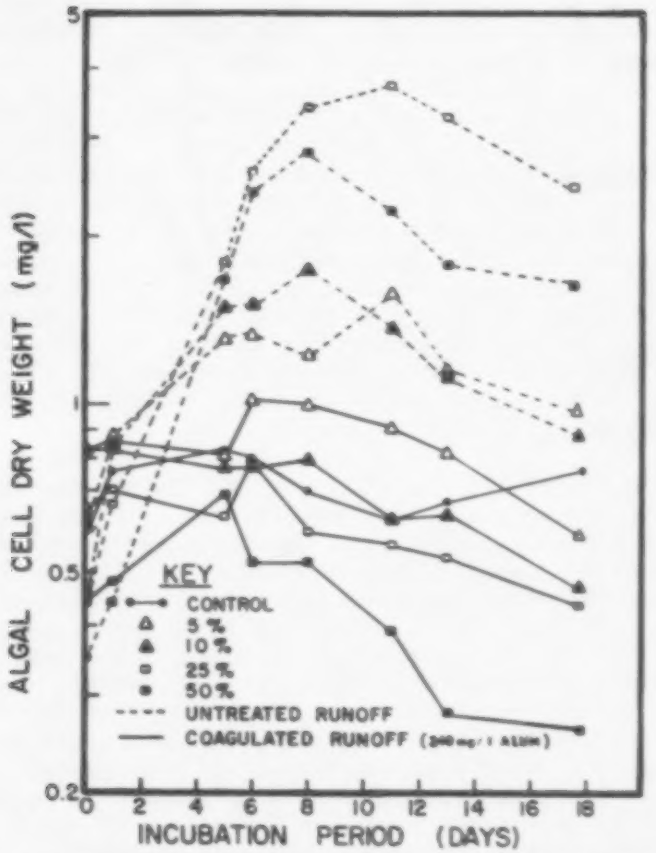


FIGURE 7 RESPONSES OF INDIGENOUS AGAL SPECIES IN LAKE EOLA TO VARIOUS CONCENTRATIONS OF STORMWATER RUNOFF AND COAGULATED RUNOFF (WATER SAMPLES COLLECTED 4-30-79, STORMWATER COLLECTED 4-28-79).

forms were no longer inhibited. Thus, it seems that coagulation of stormwater not only removes nutrients and limits algal production but it also produces a product which is less toxic to aquatic organisms.

TOXIC EFFECTS FROM STORMWATER RUNOFF

Although the algal production measured in these bioassays would certainly correspond to eutrophic conditions, it appears that production in Lake Eola may actually be partially inhibited by toxic elements in the stormwater runoff. The growth response of Selenastrum in a synthetic algal medium was measured (Harper, 1979). Since this medium is by design a phosphorus limited medium (EPA, 1978), then algal growth should continue until all available phosphorus has been utilized by organisms. A total of 120 mg of dry cell weight per liter of Selenastrum were produced from the 0.186 mg/l of phosphorus contained on the synthetic algal medium. From this relationship, it can be calculated that 0.01 mg/l of phosphorus will produce, assuming all other essential nutrients are present in sufficient quantities needed for growth, 6.45 mg/l of Selenastrum cell mass. Accordingly, the 0.017 mg/l of phosphorus present in the lake water sample collected on 8/10/78 should have produced 10.97 mg/l of dry cell mass assuming no toxic effects existed and all other nutrients necessary for growth were present. As seen from Table 6, none of the bioassay experiments with the exception of the one in which EDTA was added, were able to produce the maximum predicted yield.

The fact that the addition of EDTA was responsible for increasing the experimental yield strongly suggests the presence of heavy metal stress in Lake Eola. The additions of EDTA to a lake water sample collected on 2/21/79 resulted in a two-fold increase in cell yield (Table 6). An inhibition of 47% was calculated for this experiment between control flasks incubated with and without EDTA. EDTA may act to stabilize ferrous iron, increasing the availability of this ion for aquatic growth while at the same time suppressing heavy metal toxicity by chelation (Miller, et. al., 1976). However since the addition of iron to this experiment in control flasks without EDTA did not stimulate algal production, it was concluded that the increase in production caused by EDTA was due largely to suppression of heavy metal toxicity. Toxic levels of selected heavy metals to Selenastrum along with average Lake Eola water and stormwater levels are listed in Table 10. It can be seen that normal background concentrations of copper and zinc in Lake Eola itself are sufficient to cause continuous incipient inhibition of algal production in certain species. Average heavy metal concentrations in stormwater collected near Lake Eola indicate an algicidal level of copper with a zinc concentration sufficient to cause complete inhibition. Although no information is listed for toxic levels of lead, the average concentration of 1500 μg/l of lead found in stormwater runoff at Lake Eola can be assumed to exhibit almost certain algicidal effects. Experiments reported by Shiroyama (1976) and Greene (1975) on the toxicity of zinc to Anabaena and Selenastrum indicate that a zinc concentration corresponding to the average Lake Eola concentration of 0.048 mg/l was sufficient to reduce the yield of Anabaena and Selenastrum to only 20-40% of the maximum yield. Similiar concentrations of lead and arsenic are also present in Lake Eola water and may exhibit similar inhibitory effects. Synergistic effects due to the simultaneous presence of various heavy metals may increase this toxic effect and further limit algal growth.

TABLE 10

TOXIC LEVELS (μg/l) OF HEAVY METALS TO SELENASTRUM ALONG WITH AVERAGE LAKE EOLA WATER AND STORMWATER CONCENTRATIONS

Element	Incipient Inhibition	Complete Inhibition	Algicidal	Lake Eola	Stormwater
Cu	50	90	300	58	513
Zn	30	120	700	35	423
Cd	50	80	650	1	25
Pb	--	---	---	69	1,580
Cr	--	---	---	29	46
As	--	---	---	58	94
Ni	--	---	---	8	30

SOURCE: L.Bartlett and F. Rabe, "Effects of Copper, Zinc and Cadmium on Selenastrum Capricornutum," Water Research, Volume 8, March 1974, pp. 179-185.

LAKE MANAGEMENT PLAN

A plan for restoration of Lake Eola had been developed and supported by the US EPA, Florida DER, City of Orlando, and College of Engineering, UCF. The plan was based on findings from the previous studies and the essential elements are summarized as follows:

1. Reduction of nutrient input, especially phosphorus, by diversion of stormwater runoff, storage and/or biological uptake.
2. Inactivati~~on of~~ phosphorus release from bottom sediments in areas of the lake deeper than 4.0 meters, due to anoxic conditions. Studies are currently underway to evaluate the possibility of using water treatment sludges for inactivation.
3. Coagulation of stormwater before release to a limited area of the lake on a pilot scale for experimentation.

This plan is currently considered for implementation and a continuous monitoring system was set to measure the nature and extent of lake productivity changes. Detailed analysis and discussion of the proposed management plan is beyond the scope of this presentation.

SUMMARY AND CONCLUSIONS

From all indications, both visually and analytically, Lake Eola appears to be a lake in severe ecological distress. Persistent algal blooms exist virtually year round. Populations of the macroscopic algae *Chara* and the filamentous green algae, *Spirogyra*, became so dense during the summer months along the shoreline that in many instances it became very difficult to launch the small Jon Boat used in this research.

Concentrations of dissolved oxygen in Lake Eola, although usually at or above saturation near the surface, drop periodically during the spring and summer months to less than 1 mg/l in deep areas of 4.0 meters or more water column. Phosphorus from the bottom sediments was released up to a level of 250 mg/m^2 after two months of anoxic conditions.

Bioassay experiments were utilized to study the impact of stormwater runoff, untreated and coagulated on productivity of Lake Eola water. Additionally, possible toxic effects from heavy metals released to the lake have been discussed. It appears that bioassay techniques could become a useful tool provided proper experimental conditions and test organisms are selected. The following conclusions are reached:

1. Management of stormwater runoff is essential if restoration of Lake Eola is to be successful.
2. Considerable amounts of nutrients are released to Lake Eola by stormwater runoff. Most nutrients are released periodically for a few hours at a time over a period of six months during spring and summer. More than half of nutrients released are in soluble form.

3. Phosphorus is probably the most important nutrient in regulating algal production in Lake Eola. Addition of phosphorus to Lake Eola waters will result in an increase in algal cell production.
4. From bioassay experimentation, stormwater up to 25% mixture with lake water will enhance algal production. A mixture of lake water and dry weather flow from storm drains did not produce changes in algal production.
5. If stormwater runoff is coagulated with alum before its release to the lake, no significant increase in algal production will be produced.
6. Stormwater contains heavy metals and/or organic compounds which may be in concentrations toxic to algal species in lake water.

REFERENCES

APHA-AWWA-WPCF. Standard Methods for the Examination of Water and Wastewater, 14th Edition, American Public Health Association, Washington D.C., 1975.

Bartlett, Larry, Rabe, Fred W., and Funk, William H. "Effects of Copper, Zinc, and Cadmium on Selenastrum capricornutum," Water Research, Volume 8, pp. 179-185, 1974.

Chiaudani, G. and Vighi, M. "The N:P Ratio and Tests with Selenastrum to Predict Eutrophication in Lakes," Water Research, Volume 8, pp. 1063-1069, 1974.

Filip, Daniel S., and Middlebrooks, Joe E. "Evaluation of Sample Preparation Techniques for Algal Bioassays," Water Research, Volume 9, pp. 581-585, 1975.

Greene, J.C., Soltero, R.A., Miller, W.E., Gasperino, A.F., and Shiroyama T. "The Relationship of Laobratory Algal Assays to Measurements of Indigenous Phytoplankton in Long Lake, Washington," Biostimulation and Nutrient Assessment, Ann Arbor Science, Ann Arbor, pp. 93-125, 1976.

Greene, Joseph C., Miller, William E., Shiroyama, Tamotsu, and Merwin, Ellen. "Toxicity of Zinc to the Green Alga Selenastrum capricornutum as a Function of Phosphorus or Ionic Strength," Proceedings: Biostimulation and Nutrient Assessment Workshop, U.S. Environmental Protection Agency, pp. 28-42, October 16-18, 1975.

Harper, Harvey H. Ecological Responses To Urban Runoff in Lake Eola, M.S. Thesis, University of Central Florida, Orlando, FL, Dec., 1979.

Marshall, Frank. Phosphorus Interactions With Lake Eola Bottom Sediments, M.S. Thesis, University of Central Florida, Orlando, FL, March, 1980.

Miller, W.E., Greene, J.C., and Shiroyama, T. "Use of Algal Assays to Define Trace-Element Limitation and Heavy Metal Toxicity," Proceedings of the Symposium on Terrestrial and Aquatic Ecological Studies of the Northwest, Eastern Washington State College, Cheney, WA, pp. 317-325, 1976.

Schlindler, D.W., and Fee, E.J. "Experimental Lakes Area: Whole Lake Experiments in Eutrophication," Journal Fisheries Research Board of Canada, Volume 31, No. 5, pp. 937-953, 1974.

Shiroyama, T., Miller, W.E., Greene, J.C., and Shigihara, C. "Growth Response of *Anabaena flos-aquae* (Lyngb.) De Brebisson in Waters Collected from Long Lake Reservoir, Washington," Proceedings of the Symposium on Terrestrial and Aquatic Ecological Studies of the Northwest, Eastern Washington State College, Cheney, WA, pp. 267-275, March 26-27, 1976.

U.S. Environmental Protection Agency. *The Selenastrum capricornutum Printz Algal Assay Bottle Test*, National Eutrophication Research Program, Corvalis, OR, p. 82, 1978.

U.S. Environmental Protection Agency. *Methods for Chemical Analysis of Water and Wastes*, Environmental Monitoring and Support Laboratory, Cincinatti, OH, 1976.

WATER QUALITY AND BIOLOGICAL DEGRADATION IN AN URBAN CREEK

by

Robert Pitt* and Martin Bozeman**
Woodward Clyde Consultants
San Francisco, California

ABSTRACT

This preliminary report presents the initial results and conclusions from the EPA-sponsored demonstration study of the water quality and biological effects of urban runoff on Coyote Creek, near San Jose, California. This first phase included investigating various field procedures that would be most sensitive in evaluating water, sediment and biological changes in the creek as it passed through the urban area. The procedures identified as most promising are currently being used in additional Coyote Creek studies.

The report describes the characteristics of urban runoff affecting the creek, sources of urban runoff pollutants, effects of urban runoff and potential controls for urban runoff. Local urban runoff characterization information is summarized, based on a previous EPA sponsored demonstration project in the area (Demonstration of Non-Point Pollution Abatement Through Improved Street Cleaning Practices-EPA grant No. S-804432, Pitt 1979) and from the local "208" study (Metcalf and Eddy 1978). Sources of urban runoff pollutants in the study area are being investigated as an important part of the field activities of the project and include sampling runoff from many source areas (such as street surfaces, parking lots, landscaped areas, rooftops and rain).

Various short- and long-term biological sampling techniques were used to evaluate the fish, benthic macroinvertebrate and attache algae conditions at many stations in the creek, above and within the urban area. Creekwater and sediment samples were also obtained and analyzed for a broad list of parameters. In most cases, very pronounced gradients of these creek quality indicators were observed, with the urbanized portion of the creek being significantly degraded. Current additional monitoring is being conducted to identify the urban runoff control goals necessary to improve creek quality to adequate levels.

---

* Private Consultant, Route 1, Blue Mounds, WI 53517 (608) 437-3456.

** Aquatic Biologist, Woodward-Clyde Consultants, 3 Embarcadero Center, Suite 700, San Francisco, CA 94111 (415) 956-7070.

INTRODUCTION

The purpose of this paper is to discuss the information that has been assembled from a variety of urban runoff studies. Some of these data and findings have been published recently; others are not generally available and should be considered preliminary findings (i.e., subject to further review). This information is presented here to stimulate interest and discussion by those concerned with urban runoff problems.

This paper is separated into three major topics: (1) characterization of urban runoff, (2) sources of urban runoff pollutants, and (3) effects of urban runoff. Much of this information is based on a recently completed EPA project (Storm and Combined Sewer Section), Demonstration of Non-point Pollution Abatement Through Improved Street Cleaning Practices (Pitt 1979) and the initial phases of an on-going project (also for the Storm and Combined Sewer Sewer Section), Water Quality and Biological Effects of Urban Runoff on Coyote Creek (Pitt and Bozeman 1979). Information has also been obtained from other runoff projects that have also characterized the sources and problems of urban runoff and have developed potential control measures. Local planning agencies that have participated in "208" Studies (pursuant to Section 208 of the Clean Water Act [PL 92-500]) are very interested in urban runoff. Approximately 30 demonstration projects have been started throughout the country under the National Urban Runoff Program (NURP) of the EPA's Water Planning Division. These projects will examine many aspects of urban runoff pollution in various areas of the country.

Projects for the Storm and Combined Sewer Section have been conducted by Woodward-Clyde Consultants (WCC) in study areas within the Coyote Creek watershed in San Jose, California. The location of this watershed in relation to major cities in the San Francisco Bay Area is presented in Figure 1. The Coyote Creek watershed is located about 50 miles southwest of San Francisco and typically receives 10 to 20 inches of rain per year. The watershed, which covers about 400 square miles, is mostly undeveloped, open rangeland, semi-wilderness state park, and low-intensity agricultural land. About 25 square miles at the northern end of the watershed is a heavily urbanized portion of San Jose.

The main channel of Coyote Creek is about 80 miles long; the southern half lies in rugged coastal hills and the northern half in gently sloping valleys. Coyote Creek empties into the southern end of San Francisco Bay. Typical average daily flows in the northern part of the creek are less than 50 cubic feet per second. Major storm flows, however, can approach 1000 cubic feet per second; these flows are controlled by two dams.

The study area is located between the farthest downstream dam (Lake Anderson) and the first major confluence (Silver Creek) which is well within the City of San Jose. Of this 20-mile study section, approximately 5 miles are urban and 15 miles are non-urban areas. The non-urban section is characterized by relatively low-intensity agricultural or open space. Sampling stations were located in both the urban and non-urban sections of the stream (to allow for comparison).

CHARACTERIZATION OF URBAN RUNOFF

For this study, runoff water quality data obtained from the street cleaning demonstration study (Pitt 1979) were examined and analyzed by an equilibrium water chemistry computer program to estimate both the specific chemical compounds that would remain soluble and those that would settle out into receiving water sediments. This information (see Table 1) shows that most of the urban runoff pollutants are soluble and would be carried in the water columns of receiving waters. Almost all (95 percent) of the lead compounds present in urban runoff are expected to be insoluble particulates and, depending on their size, may settle out in the sewerage or the receiving water sediments.* Other parameters that were monitored will mostly be present in soluble forms.

The information presented in Table 1 and data obtained from the Santa Clara County area-wide wastewater management plan (Metcalf and Eddy 1978) were used to estimate the urban area unit pollutant yields for the study area. These estimated yields, on a pounds per acre per year basis, are presented in Table 2. The estimated yields of urban runoff affecting the monitoring stations in Coyote Creek are also shown. Non-urban stations are affected by runoff from undeveloped and agricultural areas, and urban stations are affected by runoff from those areas as well as the runoff from urban areas. The pollutant yields affecting the urban stations are all substantially greater than the quantities affecting the non-urban stations. For example, the total solids discharged into the creek in urban areas were more than one hundred times those of non-urban areas. The amounts of lead discharged at urban area stations were several thousand times those of non-urban stations.

SOURCES OF URBAN RUNOFF POLLUTANTS

A major area of uncertainty concerning urban runoff is how the contributions from pollutant sources in the watershed contribute to the outfall

---

* This was substantiated in field studies by the large concentrations of lead that were found in the urban creek sediments.

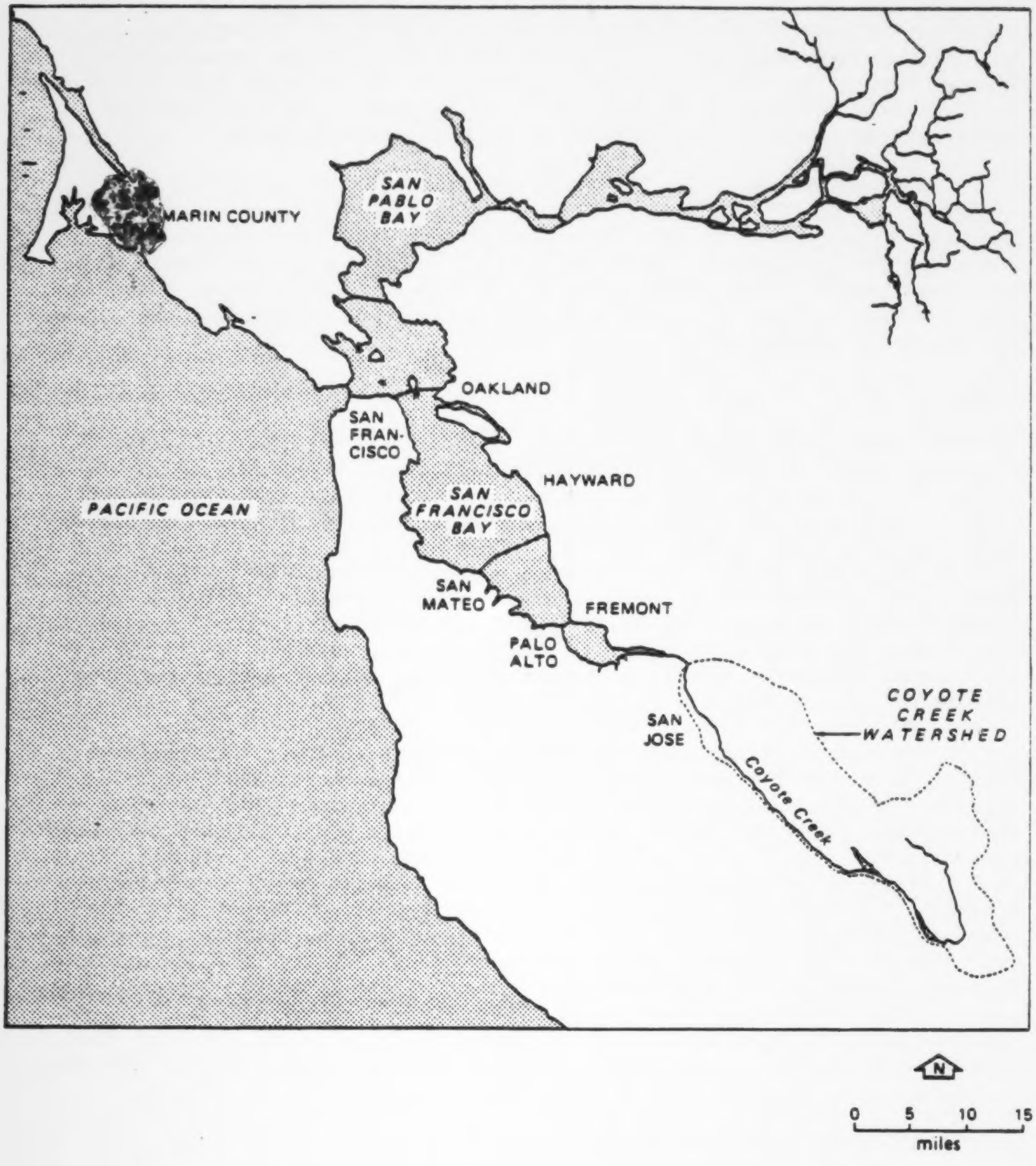


Figure 1. San Francisco Bay Area showing the general location of the Coyote Creek watershed.

Table 1. FLOW-WEIGHTED URBAN RUNOFF CONCENTRATIONS OBSERVED AND CALCULATED (mg/l, unless otherwise noted)

Parameter	Range*		Average	Parameter	Range*		Average
	Min.	Max.			Min.	Max.	
Total Ca(1)	2.8	19	13	Total Cr	0.009	0.03	0.04
Ca^{++}	2.8	18.5	12.2	$Cr(OH)_2$	<0.001	0.05	0.02
$CaSO_4$	<0.1	1.7	1.1	$CrOH^+$	<0.001	0.03	0.006
$CaHCO_3{}^+$	<0.1	0.4	0.1	Total CO_3(2)	<0.001	0.055	0.019
				Total HCO_3	<0.001	150	53
Total Mg	1.4	6.2	4.0	$HCO_3{}^-$	<0.001	0.01	0.007
Mg^{++}	1.4	5.9	3.9				
$MgSO_4$	0.1	1.1	0.6	Total SO_4	6.3	27	18
$MgHCO_3{}^+$	<0.1	0.2	<0.1	$SO_4{}^=$	6.2	25	17
Total K	1.5	3.5	2.7	Total CL	3.9	18	12
K^+	1.5	3.5	2.6	CL^-	3.9	18	12
Total Na	2.1	27	15	Total Ortho PO_4(3)	0.2	6.0	2.4
Na^+	2.1	27	15	$H_2PO_4{}^-$	0.2	4.4	1.7
Total Cu	0.013	0.05	0.03	$MgHPO_4$	<0.001	0.008	0.002
Cu^+	0.01	0.05	0.02				
Cu^{++}	<0.001	0.01	0.004	Total NO_3	0.3	1.5	0.7
$CuCO_3$	<0.001	0.01	0.005	$NO_3{}^-$	0.3	1.5	0.7
$CuHPO_4$	<0.001	0.01	<0.001				
$CuOH^+$	<0.001	0.001	<0.001	Total Hg	<0.0001	0.0002	<0.0001
Total Cd	<0.002	0.004	0.002	Kjeldahl N	3.1	15	6.7
Cd^{++}	<0.002	0.004	0.002	pH, pH Units	6.3	7.0	6.8
$CdSO_4$	<0.001	0.001	<0.001	ORP, mV	78	140	120
$CdCl^+$	<0.001	<0.001	<0.001	Temp., °C	15	15	15
Total Zn	0.10	0.32	0.18	Sec. Cond, umhos/cm	33	130	93
Zn^{++}	0.10	0.31	0.18	Turbidity, NTU	37	86	49
$ZnPO_4$	<0.001	0.003	0.001	Total Solids	110	680	300
$ZnSO_4$	<0.001	0.02	0.011	Total Dissolved Solids	34	160	91
				Suspended Solids	41	570	210
Total Pb(2,3)	0.20	0.76	0.42	Volatile Sus. Solids	5	40	23
Pb^{++}	<0.001	0.01	0.003	Dissolved Oxygen	6.5	9.9	8
$PbCO_3$	<0.001	0.02	0.01	BOD_5	17	30	24
				COD	77	350	200
				TOC	19	290	110

NOTE: The following compounds are expected to occur in urban runoff in solid form:

(1) $CaHPO_4$
(2) $PbCO_3$
(3) $Pb_3(PO_4)_2$

*The minimum and maximum range values represent individual storm flow-weighted average values. There is much larger variation when individual values during the storms are considered.

Source: Pitt and Bozeman 1979

TABLE 2. ANNUAL URBAN RUNOFF YIELD AFFECTING EACH STATION (tons/year)

Parameter	Urban Area Unit Yield lb/acre/yr	Station and Distance from Mouth of Coyote Creek (miles)*					
		Non-Urban Area			Urban Area		
		1 (38)	2 (34)	3 (28)	4 (17)	5 (15)	6 (13)
Total Solids	250	<1	<1	<10	540	660	760
COD	80	<1	<1	<10	170	210	240
BOD_5	40	<1	<1	<10	86	110	120
Kjeldahl N	2.5	<0.1	<0.1	<0.1	5.4	6.6	7.6
Ortho-PO_4	1.3	<0.01	<0.01	<0.1	2.8	3.5	4.0
Lead	1.0	<0.01	<0.01	<0.01	2.2	2.7	3.0
Zinc	0.1	<0.001	<0.001	<0.01	0.2	0.3	0.3
Chromium	0.1	<0.001	<0.001	<0.01	0.2	0.3	0.3
Copper	0.14	<0.001	<0.001	<0.01	0.3	0.4	0.4
Cadmium	0.001	<0.0001	<0.0001	<0.0001	0.002	0.003	0.003

*The urban area boundary is located at about mile 22.

Source: Pitt and Bozeman 1979

Table 3. URBAN RUNOFF POLLUTANT CONCENTRATIONS FROM MAJOR AREAS (mg/1, unless otherwise noted)

Parameter	Outfall	Gutter Flow	Parking Lot	Park Puddles	Commercial Tar and Gravel Roof	Residential Composition Shingle Roof	Rain
pH, pH Units	7.8	7.5	7.0	7.3	7.5	6.5	6.4
Specific Conductance, (umhos/cm)	185	130	45	2400	155	11.2	10.4
Turbidity, NTU	29	100	26	21	1	<1	<1
Total Solids	162	235	340	2140	186	18	30
BOD_5	8	13	22	3	7	3	3
COD	97	172	176	69	131	19	12
$O-PO_4$	0.23	0.12	0.47	0.32	0.02	0.08	0.03
Total PO_4	0.34	0.31	0.49	0.42	0.07	0.10	0.03
Kjeldahl N	1.52	2.41	1.47	1.32	4.37	0.71	0.64
NH_3	0.25	0.42	0.35	1.23	1.06	0.50	0.36
NO_3	0.74	0.42	0.13	285	0.22	0.09	0.09
S	4	2	<1	15	5	<1	<1
SO_4	13	7	<1	38	21	<1	<1
As	<0.01	<0.01	0.02	0.10	<0.01	0.01	<0.01
Zn	0.06	0.14	0.23	0.01	0.08	0.18	0.04
Pb	0.08	0.67	1.09	0.035	0.019	0.017	<0.01
Cr	0.009	0.049	0.071	0.010	<0.005	<0.005	<0.005
Cu	0.013	0.029	0.046	0.031	0.11	<0.005	0.010
Total Coliform Bacteria (MPN/100ml)	>2400	>2400	540	49	170	<2	8
Fecal Coliform Bacteria (MPN/100ml)	>2400	920	350	49	9	<2	2
Fecal Strep. Bacteria (MPN/100ml)	>2400	>2400	>2400	920	17	920	<2
Fecal Coliform/Fecal Strep. Ratio	-	<0.4	<0.2	0.5	0.5	<0.002	>1

Source: Pitt and Bozeman 1979

yield. Sources that are farther from the storm drainage system and require overland flow have a very low yield when compared with parking lots or street surfaces that are impervious and located adjacent to drainage systems. Preliminary results from the Coyote Creek project (Pitt and Bozeman 1979) which examined potential sources of urban runoff pollutants and involved the collection of runoff samples during rainstorms from small areas within San Jose (e.g., building roofs, parking lots, gutter flows) are presented in Table 3. In that study, rainfall and outfall samples were collected for chemical analyses. As expected, in almost cases rain had the lowest pollutant concentrations while parking lots and gutter flows had the greatest. Puddles that were monitored in a city park had much greater concentrations of total solids, nitrates, and higher values of specific conductance exhibited than any of the other samples.

Table 4 is a generalization of common urban runoff pollutant sources. No particular source is expected to contribute significant quantities of all the pollutants, but some of the sources are expected to be important contributors. For example, street surfaces contribute significant amounts of heavy metals; oxygen-demanding materials and nutrients are thought to come mostly from landscaped areas and vacant land.

Table 5 is a generalization which shows the pollutant contributions to major urban surfaces and the approximate delivery yields to the outfall. Vacant land and landscaped areas are among the most pervious surfaces in urban areas and are located farthest from urban drainage systems. Therefore, they contribute little flow and only a small fraction of the pollutants in urban drainage systems. For example, landscaped areas and vacant land comprise almost half of the land surface in the San Jose study area, but only about 5 percent of the rain that falls on these surfaces contributes to the outfall flow. Similarly, very little of the potential pollutant yield from these surfaces is expected to affect the outfall.

Rooftops, which comprise 15 to 20 percent of surface areas in the San Jose study area, are also located relatively long distances away from the storm sewerage system. Rooftops are impervious; however, most of the roof drainage systems in San Jose are not directly connected to the storm sewerage system and require considerable overland flow. Therefore, the outfall runoff yield from rooftops is expected to be only about 30 percent.

Sidewalks, which comprise about 5 percent of the San Jose study area, are located closer to the storm drainage system, but some of their drainage flow is directed towards adjacent landscaped or other pervious areas. Therefore, only about half of the runoff yield from sidewalks enters the receiving

Table 4. POTENTIALLY SIGNIFICANT SOURCES OF URBAN RUNOFF POLLUTANTS

	Rooftops	Street Surfaces	Parking Lots	Landscaped Areas	Vacant Land	Construction Sites	Other (Industrial and Solid Waste Runoff)
Sediment		X			X	X	
Oxygen Demanding Matter				X			
Nutrients				X	X		
Salts							
Bacteria					X		
Heavy Metals		X	X				
Pesticides/Herbicides				X		X	
Oils and Grease	X	X	X				
Floating Matter				X			
Other Toxic Materials	X	X					X

Table 5. POLLUTANTS CONTRIBUTED BY VARIOUS URBAN SOURCES

Pollutant Sources	Lawn and Landscaped Areas	Vacant Lots	Rooftops	Sidewalks	Parking Lots	Street Surface
Dustfall	X	X	X	X	X	X
Precipitation	X	X	X	X	X	X
Tire Wear	X (Adjacent)			X	X	X
Auto Exhaust Particulates	X (Adjacent)			X	X	X
Other Auto Use (Fluid Drips, Wear Prod.)				X	X	X
Vegetation Litter	X	X		X	X	X
Construction Erosion		X				
Other Litter		X		X	X	X
Bird Feces	X	X	X	X		
Dog Feces	X	X				X
Cat Feces	X	X				
Fertilizer Use	X					
Pesticide Use	X					
De-Icing Use				X	X	X
Proportion of Urban Surface Area	40%	10%	15-20%	5%	7%	15-20%
Delivery Yield to Outfall*	5%	5%	30%	45%	50%	75%

* This is the portion of the total pollutant load that leaves the surface and eventually arrives at the outfall (e.g., 45% of the pollutants which are washed off sidewalks get to the outfall; 55% is "lost" enroute and cannot be accounted for.

waters. Parking lots, which comprise 7 percent of the San Jose study area, are mostly paved and impervious. Again, some of the runoff from the parking lots (especially at homes and apartments) is directed towards adjacent impervious areas, and only about half of the parking lot runoff is expected to reach the receiving waters.

Street surfaces, however, are located very close to the storm drainage system and are almost impervious. They comprise about 15 to 20 percent of the study area, and most of the runoff from the street surfaces is expected to reach the outfall. However, some of the street surface flow does not reach the outfall, possibily because of evaporation and infiltration through streets in poor condition.

Dust fall and precipitation affect all of these major components. Although not a major pollutant source, dust fall is a mechanism for transport of pollutants. Most of the dust monitored in urban areas is resuspended particulate matter from street surfaces or wind erosion products from vacant areas. Some point source air pollutant emissions also contribute to dustfall pollution, but the bulk is contributed by other sources already considered (e.g., street dirt and other fugitive dusts).

Automobile tire wear is a substantial source of zinc in urban runoff and is mostly deposited on street surfaces and adjacent areas. About half of the settleable particulates resulting from tire wear settles on the street and the remaining material settles within about 20 feet of the roadway. Auto exhaust particulates also contribute significant amounts of heavy metals (especially lead) and mostly affect street surfaces and adjacent areas. Other automobile pollutant contributions are associated with fluid losses from drips and spills, and by mechanical wear products. Heavy metals and asbestos are also associated with those losses. Most of these pollutants directly affect parking lots and street surfaces and some pollutant material lands on adjacent areas through wind and automobile-induced air turbulence.

Vegetation litter can be a significant pollutant component in most source areas. For example, leaf fall in San Jose is an important street surface pollutant in the autumn. Although animal feces can contribute significant quantities of nutrient and bacteria to urban areas, this pollutant source mostly affects vacant land and landscaped areas. Fertilizer, pesticide, and herbicide use is generally associated with landscaped areas, but large amounts are routinely used to manage plant growth on and adjacent to impervious surfaces. Fertilizers may be used in large quantities in road landscaping projects. In cold climates, the use of deicing materials also contributes salt and particulate materials to roadways, parking lots, and adjacent areas.

Table 6 is based on preliminary results from the Coyote Creek project and estimates the percentage contribution of the various pollutants from the different source areas studied. In most cases, rooftops contribute the least amount of pollutants while the pervious areas contribute the majority of

Table 6. CONTRIBUTION FROM VARIOUS AREAS TO OUTFALL RUNOFF YIELDS (percent)

Sources	Total Solids	COD	BOD_5	Ortho PO_4	Total PO_4	Total Kjeldahl N	NH_3	NO_3	Total S
Rooftops	2	5	2	10	5	15	20	2	~ 5
Landscaped Areas and Vacant Land	50	50	50	20	15	25	20	75	50
Parking Lots	10	15	20	30	20	10	10	1	<1
Street Surfaces and Sidewalks	40	25	25	45	65	50	50	20	45

Sources	SO_4	As	Zn	Pb	Cr	Cu	Total Coliform Bacteria	Fecal Coliform Bacteria	Fecal Strep Bacteria
Rooftops	5	10	20	<1	<1	10	<1	<1	<30
Landscaped Areas and Vacant Land	40	70	1	<1	2	10	<1	<1	<3
Parking Lots	<1	15	20	20	20	20	<5	<3	>10
Street Surfaces and Sidewalks	50	<1	60	80	80	60	>95	>95	>60

solids, oxygen demanding materials, and some nutrients. Parking lots, street surfaces, and sidewalks are expected to contribute some nutrients as well as a proportion of heavy metals and bacteria, to the total outfall runoff yield.

Local erosion products and some contaminants from motor vehicles account for most of the street surface dust and dirt material (by weight). Where streets are in good repair, only minor contributions are made by wear of the street surfaces. The specific makeup of street surface contaminants is a function of many site conditions and varies widely. Many pollutant sources are specific to particular areas and on-going activities. For example, iron oxides are associated with welding operations, and strontium (used in the production of fireworks and flares) would probably be found on streets in greater quantities around holiday times or at the scenes of traffic accidents.

Relative deposition values for the different pollutant sources are summarized in Table 7. These values represent the percentage of the total pollutants deposited in urban areas; however, the amounts deposited exceed the pollutant yields to the outfall. In comparison, Table 8 shows the relative yields from these pollutant sources to the total outfall runoff yield. The deposition rates of some pollutants are relatively high for some impervious areas, but these source yields are reduced substantially when infiltration is considered. Automobile activity is responsible for most of the heavy metal yield in the runoff and about half of the total solids yield. Vegetation sources contribute most of the oxygen-demanding materials, while animal feces and fertilizers are thought to contribute most of the nitrogen in urban runoff.

A summary of predicted San Jose urban runoff characteristics is presented in Table 9. Conditions vary substantially for other areas of the country but these data do point out some important aspects of urban runoff. Of all the total solids deposited in urban areas, only about one-third will reach the outfall. Only about 10 percent of the nutrients and oxygen-demanding materials deposited would affect the receiving water quality, but most of the heavy metals deposited in the area would affect the receiving waters. The remaining pollutants that are washed off the source areas and do not reach the outfall would be accumulated in other areas of the urban environment. The most significant pollutant "sinks" in urban areas are soils, groundwater, and plants. For example, many studies have shown significant concentrations of heavy metals in roadside soils and vegetation (Farmer and Lyon 1977; McMullen and Faoro 1977; Olson and Skogerboe 1975; Pitt and Amy 1973). Deicing salts have also been shown to affect shallow groundwater near urban areas. Much of this material (about 15 percent of the total deposition of total solids) can be associated with dust fall. Most of this dust fall, however, consists of resuspended particulates from streets and vacant areas and, except for point source air pollution emissions that may settle out, is not an actual source of urban runoff pollutants.

Table 7. RELATIVE ANNUAL DEPOSITION VALUES FOR POLLUTANTS FROM VARIOUS SOURCES (percent)

Source	Total Solids	BOD_5	TKN	Pb	Zn	Cr	Cu
Precipitation	10%	3%	5%	<1%	15%	<1%	10%
Tire Wear	5	<1		10	80	10	5
Auto and Street Use	20	1	1	90	1	70	50
Vegetation	40	95	15				
Construction Erosion	20		<1	<1	2	10	2
Bird Feces	2	<1	<1	<1	<1		
Dog Feces			40				
Cat Feces			<1				
Fertilizer Use			40				
Other	3	1				10	30

Table 8. RELATIVE POLLUTANT YIELDS OF TOTAL OUTFALL RUNOFF (percent)

Source	Total Solids	BOD_5	TKN	Pb	Zn	Cr	Cu
Precipitation	10%	10%	20%	<1%	20%	<1%	5%
Tire Wear	5	1		1	80	3	2
Auto and Street Use	50	10	10	100		90	80
Vegetation	30	70	10				
Construction Erosion	5		<1	<1	<1	1	<1
Bird Feces	5	<1	1	<1	<1		
Dog Feces			30				
Cat Feces			<1				
Fertilizer Use			30				
Other	3	5				5	15

Table 9. EXPECTED SAN JOSE URBAN RUNOFF CHARACTERISTICS

	Total Solids	BOD_5	TKN	Pb	Zn	Cr	Cu
Expected Outfall Runoff Yield (lb/acre/yr)	250	20	3.5	1.0	0.15	0.1	0.15
Percentage of Deposition Yield to Outfall (%)	30	10	10	90	20	70	60
Major Sources Contributing to Runoff Yield	Auto and Street Use	Vegetation	Dog Feces and Fertilizer Use	Auto and Street Use	Tire Wear	Auto and Street Use	Auto and Street Use

EFFECTS OF URBAN RUNOFF

This section summarizes the field and laboratory activities conducted to examine the water quality and biological effects of urban runoff on Coyote Creek. Preliminary work involved sampling Coyote Creek during the fall of 1977 and spring and early summer months of 1978. The objectives were to characterize (on a macro-scale) sediment quality and biological conditions as the creek passed through urban areas. The types and magnitude of changes in the sediment quality and biological community in Coyote Creek were examined as the creek passed through San Jose* between Lake Anderson and the confluence of Silver Creek. With the exception of urban runoff, in this reach of Coyote Creek there are no pollutant discharges. Each sampling area included a stretch of stream several hundred yards long.

The following factors were analyzed in this study:

- runoff (various water quality parameters)
- sediment (BOD_5, COD, TOC, volatile solids, NH_3, NO_3, organic N, ortho PO_4, SO_4, complete elemental scan by SSMS including heavy metals, complete organic scan by MSGC including pesticides and PCB's, and particle size distribution)
- tissue analyses (for lead and zinc)
- fish
- benthic organisms (including aquatic insects, crustaceans, and molluscs)
- attached algae
- rooted aquatic vegetation (where present).

While the sediment and water column samples were only collected once, most of the biological sampling was conducted in two consecutive periods during the spring and early summer. The following paragraphs briefly describe some of the water quality and biological changes observed in Coyote Creek.

---

* This urban area includes the Keyes Street and Tropicana study areas that were investigated as part of the San Jose street cleaning demonstration project.

Runoff Water Quality

The ranges and average runoff pollutant concentrations for several monitored storms in San Jose were presented in Table 1. In the urban areas, Coyote Creek had water quality concentrations similar to those of runoff during storms. These water quality conditions only occur during storm runoff; the water quality in the urban areas of Coyote Creek improves somewhat during dry weather.

In Table 10, observed runoff concentrations are compared with recommended water quality criteria for various beneficial uses. The water quality criteria values for these uses are recommended maximum limits designed to protect the beneficial uses with a reasonable amount of safety. If a monitored concentration exceeds these criteria, it does not mean that a problem exists but that one could occur, and that additional monitoring may be necessary (to define the relationships between water quality and impairment of the beneficial uses for the specific receiving water). Beneficial uses that are actually impaired by urban runoff must be carefully considered. For example, no one would swim in the creek during storms, but aquatic life could be severely impacted (intermittently) during storms.

The following list summarizes the parameters that exceeded the recommended beneficial use criteria:

Livestock: Pb*
Wildlife: None
Aquatic life: Cr, Cd*, Pb*, Hg*, BOD_5, turbidity*, suspended solids*
Marine life: PO_4*, Cd, Cu, Zn
Recreational uses: PO_4*,
Freshwater public supply: Cd, Pb*
Irrigation: Cd.

These data indicate that those parameters most potentially responsible for water quality impairment are solids, cadmium, lead, and mercury for aquatic life uses; orthophosphates for marine life; orthophosphates for eutrophication; and lead for freshwater public supply.

Limited data are available concerning the water quality of Coyote Creek in urban areas during dry weather and in non-urban areas during wet weather. These data are summarized in Table 11 and indicate that the high suspended solids and phosphate concentrations in non-urban areas during wet weather can greatly exceed the aquatic life and marine life criteria. The phosphate concentrations may also exceed the criteria for recreation.

---

*The maximum monitored value was greater than 10 times the minimum recommended criterion.

Table 10. RUNOFF WATER QUALITY COMPARED TO BENEFICIAL USE CRITERIA

Parameter[a]	Overall Observed Range	Overall Observed Average	Beneficial Use Criteria[b]						
			Irrigation	Livestock	Wildlife	Aquatic Life	Marine Life	Recreational Uses	Freshwater Public Supply
pH (pH units)	6.0 → 7.6	6.7	4.5 → 9.0 desired	—	6.0 → 9.0 desired	6.0 → 9.0 desired	6.5 → 8.5 desired	5.0 → 9.0 desired	5.0 → 9.0 desired
Temp. (°C)	14 → 16.5	16	Narrative	—	Maintain natural pattern	Narrative	Narrative	86°F	Narrative
DO	5.4 → 12.8	8.0	—	—	—	Usually 5.0 mg/l min.	6.0 mg/l min.	—	Narrative
Turbidity (NTU)	4.8 → 130	49	—	—	—	Small change	—	4 ft (secchi)	Narrative
TDS	22 → 376	150	500 → 5000 mg/l max.	—	—	Narrative	—	—	Narrative
SS	15 → 845	240	Narrative	—	—	80 mg/l	—	—	—
NO_3	0.3 → 1.5	0.7	Narrative	450 mg/l (including NO_2)	—	—	—	—	45 mg/l
PO_4	0.2 → 17.6	2.4	—	—	—	—	0.0003 mg/l	0.3 mg/l for streams; 0.08 for lakes	Narrative
Cl	3.9 → 17.6	12	—	—	—	—	—	—	250 mg/l
SO_4	6.3 → 27	18	—	—	—	—	—	—	250 mg/l
Na	2.1 → 26.8	15	Narrative	—	—	—	—	—	Narrative
Cd	<0.002 → 0.04	0.01	0.01 → 0.05 mg/l max.	50 mg/l	—	0.004 → 0.03 mg/l max. for soft → hard water	0.01 mg/l	—	0.01 mg/l
Cr	0.005 → 0.04	0.02	0.1 → 1.0 mg/l max.	1.0 mg/l	—	0.03 mg/l	0.1 mg/l	—	0.05 mg/l
Cu	0.01 → 0.09	0.03	0.02 → 5.0 mg/l max.	0.5 mg/l	—	Narrative	0.05 mg/l	—	1 mg/l
Pb	0.10 → 1.5	0.4	5.0 → 10.0 mg/l max.	0.1 mg/l	—	0.03 mg/l	Narrative	—	0.05 mg/l
Hg	<0.0001 → 0.0006	<0.0001	—	0.001 mg/l	Narrat-rative	0.00005 mg/l	0.1 mg/l	—	0.002 mg/l
Zn	0.06 → 0.55	0.18	—	25 mg/l	—	Narrative	0.1 mg/l	—	5 mg/l
BOD_5	17 → 31	24	—	—	—	10 mg/l	Narrative	—	Narrative

Sources: McKee and Wolf 1963; USEPA 1973; USEPA 1975.

[a] Parameters are measured in mg/l unless otherwise noted.
[b] Maximum limits unless stated as desired range or minimum values.

Table 11. WATER QUALITY CONDITIONS IN COYOTE CREEK BY LOCATION AND WEATHER CONDITION

Parameter (mg/l, unless otherwise noted)	Urban Area		Non-Urban Area	
	Wet Weather	Dry Weather	Wet Wether	Dry Weather
pH (pH Units)	6.7	7.8	-	7.9
Temperature (°C)	16	15	-	15
Ca	13	-	-	40
Mg	4.0	-	-	25
Na	0.01	-	-	19
K	2.7	-	-	1.9
HCO_3	54	130	-	170
CO_3	0.019	<24	-	3.0
SO_4	18	55	-	37
Cl	12	60	-	15
Total Hardness	-*	250	-	200
Total Alkalinity	-	240	-	140
Total Solids	310	-	-	-
TDS	150	-	-	280
Sus. Solids	210	-	600	-
Vol. Sus. Solids	23	-	90	-
Turbidity (NTU)	49	14	-	18
Spec. Cond. (umhos/cm)	160	520	-	400
Dissolved Oxygen	8.0	7 .6	-	11
BOD_5	24	-	4	-
COD	200	-	-	-
Kjeldahl N	7	-	2	-
NO_3	0.7	0.84	-	1.2
NO_2	-	0.024	-	<0.002
NH_3	-	0.95	-	0.33
$O-PO_4$	2.4	-	0.8	-
TOC	110	-	-	0.6
Pb	0.4	-	-	-
Zn	0.18	-	-	-
Cu	0.03	-	-	-
Cr	0.02	-	-	-
Cd	<0.002	-	-	-
Hg	<0.0001	-	-	-
Total Coliform Bacteria (MPN/100ml)	>2400	>1900	-	>1300
Fecal Coliform Bacteria (MPN/100ml)	>2400	-	-	-
Fecal Strep. Bacteria (MPN/100ml)	>2400	-	-	-

* The blanks signify where no data is shown.

A comparison between secondary sanitary wastewater effluent concentrations and urban runoff concentrations for the study area is presented in Table 12. The average and peak 1-hour runoff concentrations and the average secondary sanitary wastewater effluent concentrations are shown. This sanitary wastewater treatment facility is a modern, advanced, secondary treatment plant for the study area. The short-term effects of urban runoff on receiving waters occur (by definition) during and immediately following a runoff event (i.e., short-term effects are associated with instantaneous concentrations). A comparison between urban runoff average concentrations and the sanitary wastewater treatment plant effluent average concentrations shows that the concentrations of lead, suspended solids, BOD_5, COD, TOC, turbidity, zinc, chromium, and cadmium are all higher in the runoff than in the sanitary wastewater effluent. In addition to the previously listed parameters, copper and Kjeldahl nitrogen have greater runoff peak concentrations than the average wastewater concentrations. Therefore, urban runoff may have more important short-term effects on receiving waters than average treated sanitary wastewater effluent concentrations.

Annual yields of these sources provides an indication of potential long-term problems. Table 12 also shows the annual sanitary wastewater treatment plant effluent yield expressed as tons per year (derived from monthly average concentrations and effluent quantities) and the calculated street surface portion of the annual urban runoff yield expressed in tons per year for a similar service area. On an annual basis, the total orthophosphates and Kjeldahl nitrogen yields associated with street surface runoff alone are less than 4 percent of the total yield from sanitary wastewater treatment plant effluent and street surface runoff. Total solids, cadmium, and mercury in the street surface runoff contribute from 5 to 10 percent of the total yield (respectively), while COD, BOD_5, and copper contribute from 10 to 50 percent of this total yield. Suspended solids, chromium, zinc, and lead in the street surface runoff contribute more than 50 percent of the total yield.

These data show that, for receiving waters in which both secondary treated sanitary wastewater and untreated urban runoff are deposited, additional improvements in the sanitary wastewater effluent may not be as cost-effective as some urban runoff treatment (except for nutrients). That would be especially true for lead, where more than 95 percent of the total wasteload is from street surface runoff. For example, if all of the lead were removed from the sanitary wastewater effluent, the total annual lead discharge would only decrease by about 4 percent.

Grab samples of Coyote Creek water were analyzed for major parameters during dry weather. The water was very turbid, hard to very hard, and had high ammonia and coliform bacteria concentrations. Figure 2 shows a marked increase in nitrites, ammonia, turbidity, chlorides, and specific conductance as the creek passed through urban areas of the watershed. The concentrations are expected to be greater and the trends more evident during and immediately after rains, as previously shown in Table 11.

Table 12. COMPARISON OF URBAN RUNOFF AND WASTEWATER TREATMENT PLANT EFFLUENT

Parameter	Runoff Concentration (mg/l unless otherwise noted) Avg.	Runoff Concentration (mg/l unless otherwise noted) Peak (1-hr)	STP[a] Effluent Concentration (mg/l unless otherwise noted) Average	Ratio of Avg. Runoff to STP Conc.	Ratio of Peak Runoff to Avg. STP Conc.	Annual Street Surface Runoff[b] (tons/yr.)	Annual STP Effluent (tons/yr.)	Ratio of Street Surfac Runoff to STP Annual Yield
Ca^{++}	13	19	65	0.20	0.29	290	8000	0.04
K^+	2.7	3.5	24	0.11	0.15	60	3200	0.02
Mg^{++}	4.0	6.2	35	0.11	0.18	90	4700	0.02
Na^+	15	27	220	0.07	0.12	330	30,000	0.01
Cl^-	12	18	330	0.04	0.05	270	45,000	0.00
$SO_4^=$	18	27	150	0.12	0.18	430	20,000	0.02
HCO_3	54	150	230	0.23	0.66	1200	32,000	0.04
NO_3	0.7	1.5	4.9	0.14	0.31	17	660	0.03
BOD_5	24	30	21	1.1	1.4	620	2800	0.2
COD	200	350	35[d]	5.6	10	2500	4700[d]	0.5
KN	6.7	25	24	0.28	1.1	110	3200	0.03
Ortho PO_4	2.4	18	19	0.13	0.92	40	2600	0.02
Total Solids	350	950	1000	0.34	0.92	7800	140,000	0.04
TDS[e]	150	380	1000	0.15	0.37	3300	140,000	0.02
Suspended Solids	240	850	26	9.2	32	5900	3500	1.7
Cd	0.01	0.04	0.002	5	20	0.03	0.27	0.1
Cr	0.02	0.04	0.016	1.3	2.5	3.1	2.2	1.4
Cu	0.03	0.09	0.081	0.37	1.1	4.7	11	0.4
Pb	0.4	1.5	0.0098	41	150	31	1.3	24
Zn	0.18	0.55	0.087	2.1	6.3	4.7	12	0.4
Hg	<0.0001	0.0006	0.0019	<0.05	0.32	<0.002	0.26	<0.008
Specific Conductance (umhos/cm)	120	660	1900	0.06	0.36	—	—	—
Turbidity (NTU)	49	130	20	2.5	6.5	—	—	—
pH (pH Units)	6.7	7.6	7.6	—	—	—	—	—
TOC[f]	110	290	30	3.5	9.7	2500	4100	0.6

[a] Secondary sanitary wastewater treatment plant serving 850,000 people.

[b] About 200 people correspond to 1-curb-mile (2,880 curb-miles in San Jose/575,000 population). Therefore, a population of 850,000 corresponds to about 4,250 curb-miles, with about 1,100 curb-miles of streets surfaced with oil and screens, and about 62,000 acres.

[c] Estimated. [d] Total dissolved solids. [e] Total organic carbon.

Source: Pitt 1979

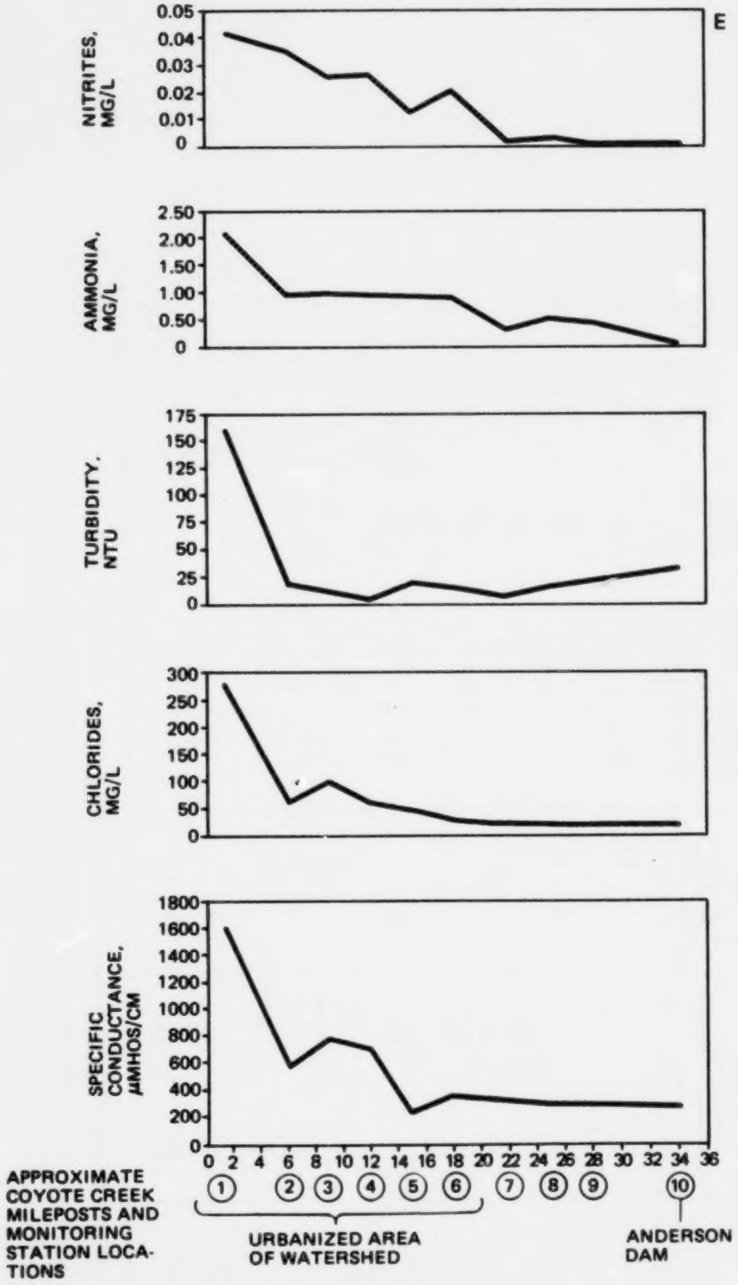


Figure 2. WATER QUALITY TRENDS ALONG COYOTE CREEK (March 31, 1977)

Sediment Analyses

Sediment samples were obtained by carefully scooping bottom material into glass jars. These jars were then sealed underwater. The samples were frozen and delivered to a laboratory for analyses. As shown in Figures 3 and 4 and Table 13, samples from urban areas contained higher concentrations of many of the parameters in comparison with samples from non-urban areas. Sulfates (33 to 60 times greater), lead (10 times greater), arsenic (9 times greater), BOD_5 (up to 4.4 times greater) and orthophosphates (up to about 4 times greater) are examples of these higher concentrations. In addition, much more silt was found in samples from urban areas. The urban samples also had significantly greater concentrations of high molecular weight oxygenated hydrocarbon compounds.

Tissue Analyses

Selected samples of fish (*Gambusia affinis*), filamentous algae (*Cladophora* sp.), crayfish (*Procambarus clarkii*), and cattail plant segments (*Typha* sp.) were collected from both the non-urban and urban reaches of Coyote Creek. Each sample was chemically digested and analyzed for whole organism lead and zinc concentrations. As shown in Tables 14 and 15, lead concentrations in urban samples of algae, crayfish, and cattails were 2 to 3 times greater than in non-urban samples. Zinc concentrations in urban algae and cattail samples were about 3 times those of the non-urban sample concentrations. Fish, lead, and zinc concentrations did not increase noticeably. Concentrations of lead and zinc in the organisms were greater than the concentrations in the sediments for many of the samples and stations (up to a maximum factor of about six). The accumulation of zinc in the crayfish, attached algae, and fish was at least 300:1 when compared with zinc concentrations in the water. The accumulation of lead in attached algae in the urban areas of Coyote Creek was at least 500:1. For crayfish, the accumulation of lead was at least 100:1 compared to lead concentrations in the water.

Biological Characterization

Fish, benthic macroinvertebrates, and attached algae were collected from representative habitats throughout the study area. Seines were used to capture fish from both riffles and pools. Captured fish were identified and counted and the total length of each individual was recorded. Replicate benthic macroinvertebrate samples were collected from natural substrates in both pool and riffle habitats with an Ekman dredge and Surber sampler. In addition, artificial substrates (replicate pairs of Hester-Dendy multiplate samplers) were used at each sampling area. The benthic samples were washed through a 500μ sieve, and the organisms retained on the screen were picked, sorted, and preserved in a 70-percent ethanol solution for later identification and enumeration. Attached algae were sampled from both natural and artifical substrates throughout the various reaches of the stream. Qualitative samples of attached algae were collected by scraping uniform areas of

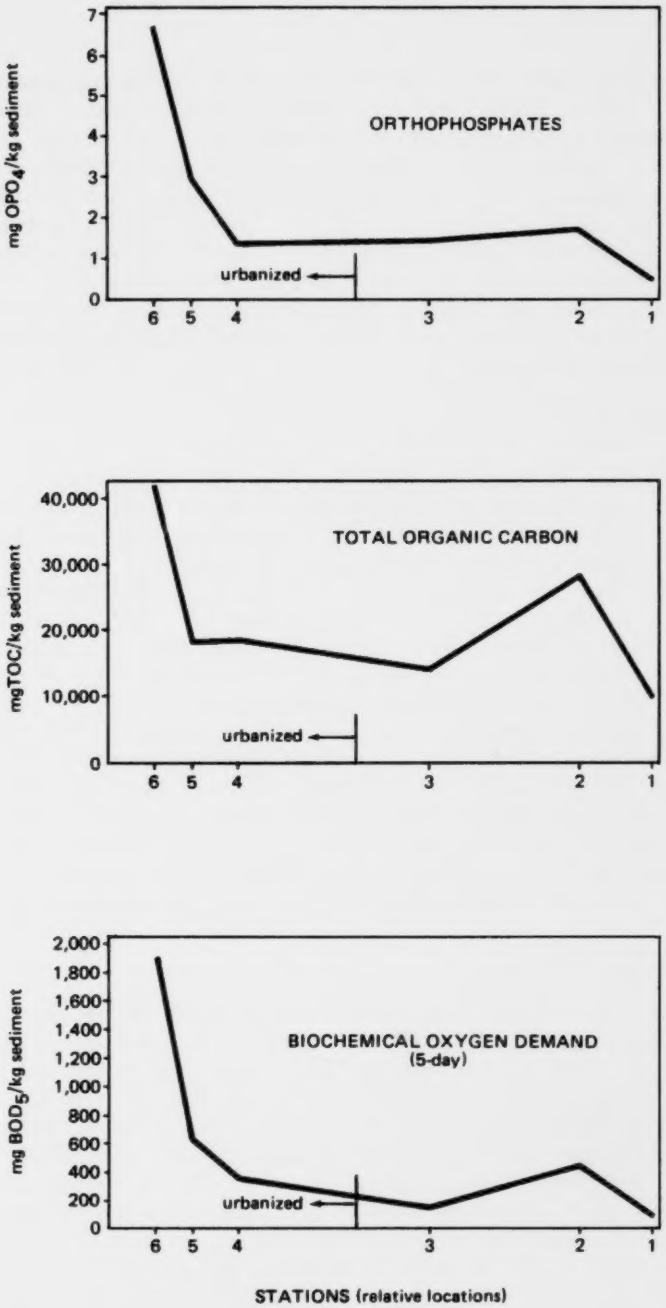


Figure 3. SEDIMENT QUALITY CONDITIONS
ALONG COYOTE CREEK

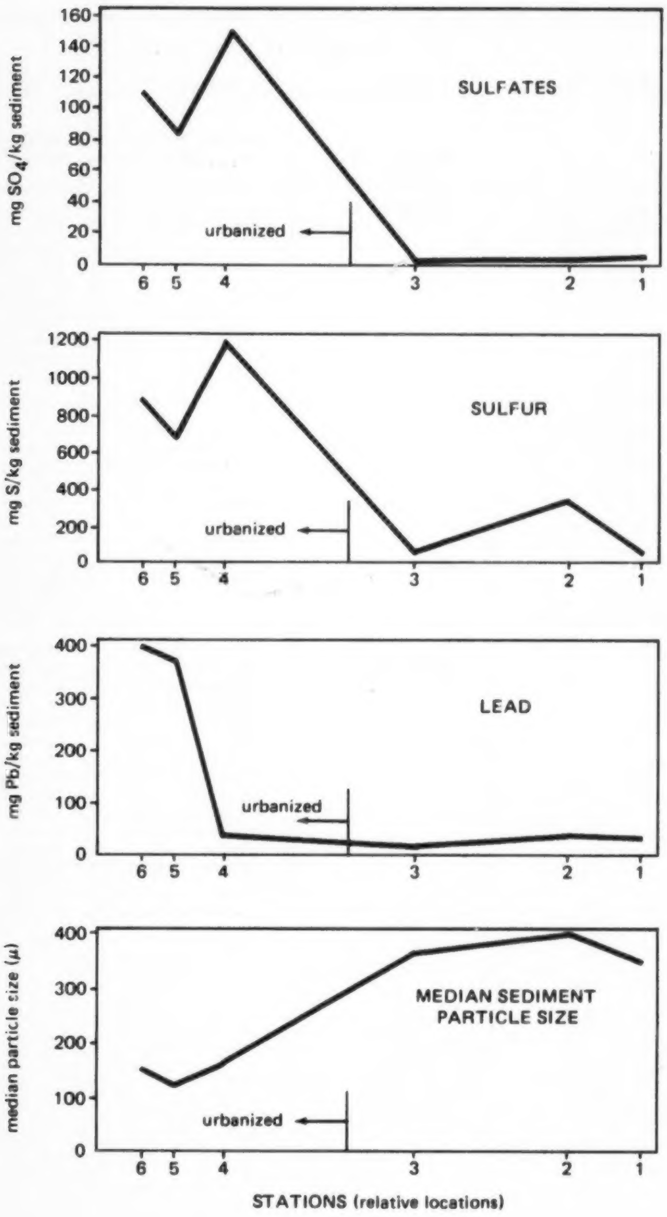


Figure 4. SEDIMENT QUALITY CONDITIONS ALONG COYOTE CREEK

Table 13. SEDIMENT CONCENTRATION INCREASES BETWEEN STATION 2 (NON-URBAN) AND DOWNSTREAM STATIONS

	Stations			
	3 (Non-Urban)	4 (Urban)	5 (Urban)	6 (Urban)
Greater than 10x Station 2 Values		Sulfate (60)	Sulfate (33) Lead (10)	Sulfate (44) Lead (11)
Between 3.0 and 10x Greater than Station 2 Values	Nickel (8.0) Chromium (6.5)	Sulfur (3.8)	Arsenic (8.7) Hafnium (4.7)	Arsenic (8.7x) Thallium (4.8) Hafnium (4.7) BOD_5 (4.4) Praseodynium (4.4) Ortho PO_4 (3.9) Silver (>3.5) Erbium (3.5) Ytterbium (3.5) Tantalum (3.0)
Between 2.0 and 2.9x Greater than Station 2 Values	Cobalt (2.9) Manganese (2.6) Tantalum (2.3)	Hafnium (2.4) Chromium (2.0) Ytterbium (2.0)	Gallium (2.4) Tantalum (2.3) Thorium (2.3) Sulfur (2.1) Antimony (2.1) Niobium (2.1) Cadmium (2.0) Ytterbium (2.0) Yttrium (2.0)	Sulfur (2.7) Cadmium (2.6) Tungsten (2.6) Lanthanum (2.5) Bismuth (>2.4) Thorium (2.3) Yttrium (2.3) Antimony (2.1) Lutecium (2.1) Gallium (2.0)
Between 1.3 and 1.9x Greater than Station 2 Values	Scandium (1.9)	Erbium (1.9) Barium (1.5) Tantalum (1.3)	Chlorine (1.9) Erbium (1.8) Neodymium (1.8) Silver (>1.7) Zinc (1.7) Ortho PO_4 (1.7) Phosphorous (1.7) Europium (1.5) BOD_5 (1.5) Lanthanum (1.4) Thallium (>1.3) Holmium (1.3) Selenium (1.3)	Europium (1.9) Gadolinium (1.9) Niobium (1.9) Uranium (1.9) Chlorine (1.7) Germanium (1.7) Tin (1.7) Titanium (1.7) Mercury (1.6) TOC (1.5) Thallium (>1.3) Holmium (1.3) Selenium (1.3) COD (1.3)

Table 14. LEAD CONCENTRATIONS IN BIOLOGICAL ORGANISMS*
(mg lead/kg dry tissue)

Specimen	Non-Urban Reach Station 1	Non-Urban Reach Station 2	Non-Urban Reach Station 3	Urban Reach Station 4	Urban Reach Station 5	Urban Reach Station 6
Fish	<40	-	-	<30	<40	<50
Attached Algae	<20	<30	<30	200	170	70
Crayfish	14	-	<30	29	<36	40
Higher Aquatic Plants	<20	<30	<30	<30	<50	60
Sediment	28	37	16	37	370	400

*The lead concentration in the urban sections of Coyote Creek during storms averaged about 0.4 mg/l. Dry weather and lead concentrations from non-urban areas are expected to be much less.

Source: Pitt and Bozeman 1979

Table 15. ZINC CONCENTRATIONS IN BIOLOGICAL ORGANISMS*
(mg zinc/kg dry tissue)

Specimen	Non-Urban Reach Station 1	Non-Urban Reach Station 2	Non-Urban Reach Station 3	Urban Reach Station 4	Urban Reach Station 5	Urban Reach Station 6
Fish	135	-	-	100	120	130
Attached Algae	6.5	24	17	160	135	69
Crayfish	80	-	90	89	140	62
Higher Aquatic Plants	9	78	26	40	150	210
Sediment	70	70	14	30	120	70

*The zinc concentration in the urban sections of Coyote Creek during storms averaged about 0.2 mg/l. Dry weather and zinc concentrations from non-urban areas are expected to be much less.

Source: Pitt and Bozeman 1979

natural substrates such as logs, rocks, etc. Quantitative collections were made with artificial substrates (diatometers equipped with glass slides) that were suspended in the water column. Qualitative samples were preserved in a 5-percent buffered formalin solution for later identification. Diatometer samples were scraped, cleaned with 30-percent hydrogen peroxide and potassium permanganate, identified, and counted.

Fish. Seine collections from the non-urban sections of the study area indicated the presence of 12 species of fish, half of which are native to the Coyote Creek system. Similar collections in the urban sections of the study area yielded only one native and three introduced fish species. As seen in Table 16, the non-urban section of the stream supports a comparatively diverse assemblage of fish that includes native species such as the California roach, hitch, Sacramento blackfish, Sacramento sucker, threespine stickleback, and prickly sculpin. Collectively, those species comprised over 60 percent of the 366 fish collected from the upper reaches of the study area. In contrast, hitch, the only native fish collected from the urban sections of the study area, represented less than 1 percent of the 1124 fish captured in the lower section of Coyote Creek. Hitch generally exhibit a preference for quiet water habitat and are characteristic of warm, low elevation lakes, sloughs, sluggish rivers, and ponds (Calhoun 1966; Moyle 1976). In streams of the San Joaquin River system in the Sierra Nevada foothills of central California, Moyle and Nichols (1973) found hitch to be most abundant in warm, sandy-bottomed streams with large pools where introduced species such as green sunfish, largemouth bass, and mosquitofish were common. Similarly, in the lower portions of Coyote Creek hitch were found to be associated with green sunfish, fathead minnows, and mosquitofish. However, mosquitofish completely dominated the collections from the urban sections of the creek, since they represented over 98 percent of the total number of fish captured. Mosquitofish are particularly well adapted to extreme environmental conditions including those imposed by stagnant waters with low dissolved oxygen concentrations and elevated temperatures. In the foothill streams of the Sierra Nevada, Moyle and Nichols (1973) found mosquitofish to be most abundant in disturbed portions of intermittent streams, especially in warm, turbid pools.

Benthic Organisms. The taxonomic composition and relative abundance of benthic macroinvertebrates collected from both natural and artificial substrates in Coyote Creek are presented in Tables 17 and 18. The benthos in the upper reaches of the creek consists primarily of immature dipterans (midges and blackflies) as well as certain clean water taxa such as mayflies and caddisflies. The benthos of the lower reaches of the creek is dominated exclusively by pollution-tolerant oligochaete worms (tubificids).

In general, the abundance and diversity of taxa appear to be greatest in the non-urban sections of Coyote Creek. Figure 5 shows the trend of the overall decrease in the total number of benthic taxa encountered in the urban sections of the study area.

Table 16. TAXONOMIC COMPOSITION AND RELATIVE ABUNDANCE OF FISH COLLECTED IN SEINE SAMPLES FROM COYOTE CREEK DURING FALL 1977 AND SPRING 1978

	Non-Urban Reach		Urban Reach	
Species	Relative Abundance (%)	Length Range (mm)	Relative Abundance (%)	Length Range (mm)
Cyprinidae - minnows and carps				
California roach (*Hesperoleucus symmetricus*)	2.7	32 to 100	-	-
Hitch (*Lavinia exilicauda*)	3.6	78 to 292	0.4	48 to 142
Sacramento blackfish (*Orthodon microlepidotus*)	2.2	160 to 340		
Fathead minnow (*Pimephales promelas*)	3.8	27 to 66	1.2	35 to 65
Catostomidae - suckers				
Sacramento sucker (*Catostomus occidentalis*)	18.3	30 to 406	-	-
Poeciliidae - live bearers				
Mosquitofish (*Gambusia affinis*)	20.5	18 to 52	98.3	15 to 52
Gasterosteidae - sticklebacks				
Threespine stickleback (*Gasterosteus aculeatus*)	33.1	34 to 50	-	-
Centrarchidae - sunfish				
Green sunfish (*Lepomis cyanellus*)	9.0	36 to 123	0.1	30
Bluegill (*Lepomis macrochirus*)	4.4	35 to 96	-	-
Largemouth bass (*Micropterus salmoides*)	0.5	89 to 350	-	-
Black crappie (*Pomoxis nigromaculatus*)	0.5	64 to 219	-	-
Cottidae - sculpins				
Prickly sculpin (*Cottus asper*)	1.4	38 to 90	-	-
Total Number of Fish Collected	366		1124	

Source: Pitt and Bozeman 1979

Table 17. TAXONOMIC COMPOSITION AND RELATIVE ABUNDANCE OF BENTHIC MACROINVERTEBRATES COLLECTED IN THE NON-URBAN REACH OF COYOTE CREEK DURING SPRING OF 1978

	Relative Abundance (%) of Each Taxon Within The Sample								
	STATION 1			STATION 2			STATION 3		
Taxon	Ekman Dredge	Surber Sampler	Artificial* Substrate	Ekman Dredge	Surber Sampler	Artificial Substrate	Ekman Dredge	Surber Sampler	Artificial Substrate
Oligochaeta**	48.7	23.2	2.3	89.8	0.4	1.4	-	45.3	-
Hirudinea	0.7	-	-	2.2	0.4	-	-	-	-
Crustacea									
Amphipoda									
Talitridae									
Hyalella azteca	-	-	-	-	1.2	-	-	-	-
Insecta									
Ephemeroptera									
Baetidae									
Baetis sp.	-	10.8	0.9	-	4.9	5.2	-	4.7	-
Centroptilium sp.	-	-	-	-	1.2	-	-	-	23.5
Ephemerellidae									
Ephemerella sp.	-	-	0.6	-	6.1	3.8	-	15.1	-
Leptophlebiidae									
Habrophlebiodes sp.	-	4.5	-	-	-	-	-	-	-
Hemiptera									
Corixidae	-	-	-	-	1.7	-	1.1	-	-
Coleoptera									
Dytiscidae	-	-	-	-	0.4	0.8	-	4.7	-
Trichoptera									
Hydropsychidae							-	4.7	-
Cheumatopsyche sp.	-	-	-	-	-	-	-	-	-
Hydroptilidae	-	0.8	-	-	-	-	-	-	-
Diptera									
Ceratopogonidae	0.4	-	-	-	-	-	93.6	15.1	-
Chironomidae	46.8	2.7	36.0	0.7	2.4	18.7	1.1	-	-
Empididae	0.4	-	-	-	-	-	-	-	-
Muscidae	-	-	-	0.7	2.9	-	-	10.4	76.5
Simuliidae	-	56.4	59.9	-	70.2	27.9	4.2	-	-
Tabanidae	-	-	-	0.7	-	-	-	-	-
Tipulidae	-	0.8	-	-	-	-	-	-	-
Gastropoda									
Lymnaeidae									
Lymnaea sp.	0.4	-	-	0.7	0.4	-	-	-	-
Physidae									
Physa sp.	-	0.8	-	1.5	3.3	30.1	-	-	-
Planorbidae									
Promenetus sp.	0.7	-	0.3	3.7	4.5	12.1	-	-	-
Pelecypoda									
Sphaeriidae									
Pisidium sp.	1.9	-	-	-	-	-	-	-	-
Total Number of Organisms/m^2	5836	602	1428	2952	1323	555	2046	106	17

Source: Pitt and Bozeman 1979

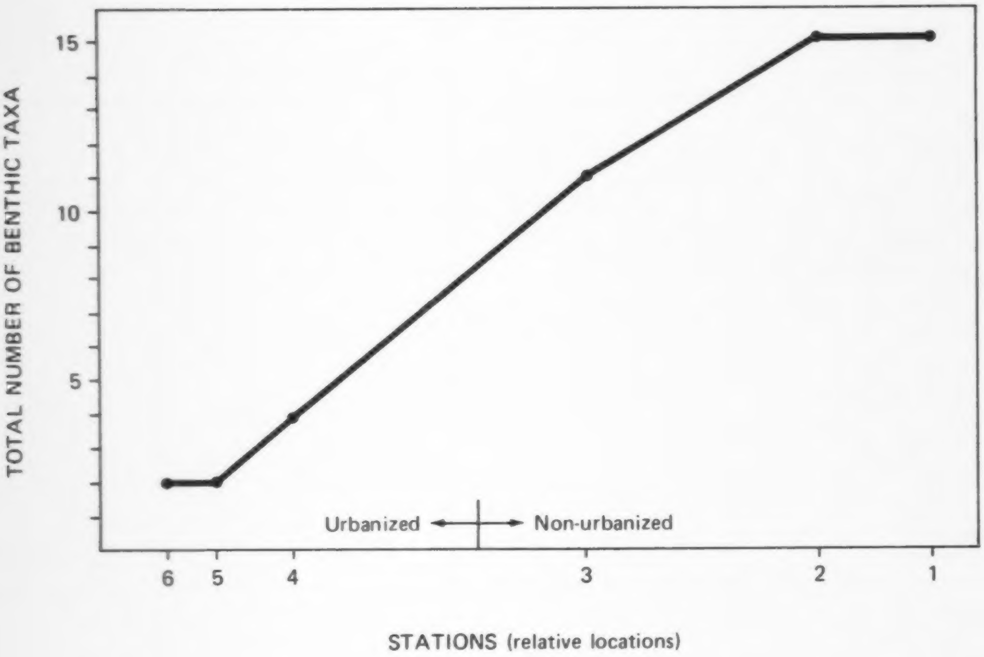
*Method of Collection at each location.
**The majority of these worms belonged to the family Lumbriculidae.

Table 18. TAXONOMIC COMPOSITION AND RELATIVE ABUNDANCE OF BENTHIC MACROINVERTEBRATES COLLECTED IN THE URBAN REACH OF COYOTE CREEK DURING SPRING OF 1978

	Relative Abundance (%) of Each Taxon Within The Sample								
	STATION 4			STATION 5			STATION 6		
Taxon	Ekman Dredge	Surber Sampler	Artificial* Substrate	Ekman Dredge	Surber Sampler	Artificial Substrate	Ekman Dredge	Surber Sampler	Artificial Substrate
Oligochaeta**	100.0	99.1	79.7	100.0	94.5	54.3	100.0	99.5	34.9
Hirudinea	-	-	-	-	-	-	-	-	-
Crustacea									
Amphipoda									
Talitridae									
Hyalella azteca	-	-	-	-	-	-	-	-	-
Insecta									
Ephemeroptera									
Baetidae									
Baetis sp.	-			-	-	-	-	-	-
Centroptilium sp.	-			-	-	-	-	-	-
Ephemerellidae									
Ephemerella sp.	-	-	-	-	-	-	-	-	-
Leptophlebiidae									
Habrophlebiodes sp.	-	-	-	-	-	-	-	-	-
Hemiptera									
Corixidae	-	-	-						
Coleoptera				-	-	-	-	-	-
Dytiscidae	-	-	-						
Trichoptera				-	-	-	-	-	-
Hydropsychidae	-	-	-						
Cheumatopsyche sp.	-	-	-						
Hydroptilidae	-	-	-	-	-	-	-	-	-
Diptera				-	-	-	-	-	-
Ceratopogonidae	-	-	15.5	-	-	-	-	-	-
Chironomidae	-	-	-	-	5.5	45.7	-	0.5	65.1
Empididae	-	-	-	-	-	-	-	-	-
Muscidae	-	0.1	4.8	-	-	-	-	-	-
Simuliidae	-	-	-	-	-	-	-	-	-
Tabanidae	-	-	-	-	-	-	-	-	-
Tipulidae	-	-	-	-	-	-	-	-	-
Gastropoda									
Lymnaeidae									
Lymnaea sp.	-	-	-	-	-	-	-	-	-
Physidae									
Physa sp.	-	-	-	-	-	-	-	-	-
Planorbidae									
Promenetus sp.	-	-	-	-	-	-	-	-	-
Pelecypoda									
Sphaeriidae									
Pisidium sp.	-	0.8	-	-	-	-	-	-	-
Total Number of Organisms/m^2	926	3432	84	1335	290	138	1787	3362	83

Source: Pitt and Bozeman 1979

*Method of Collection at each location.
**The majority of these worms belonged to the family Tubificidae.



Source: Pitt and Bozeman 1979

Figure 5. Abundance of benthic taxa collected from natural and artificial substrates in Coyote Creek during Spring of 1978.

Attached Algae. Qualitative samples from natural substrates indicated that the filamentous alga, *Cladophora* sp., was found throughout the study area. However, growth of this alga reached its greatest proportions in the upper sections of Coyote Creek. The taxonomic composition and relative abundance of diatoms collected from artificial substrates placed at each sample location are presented in Table 19. The periphyton in non-urban sections of the creek was dominated by the genera *Cocconeis* and *Achnanthes*. The genera *Nitzschia* and *Navicula*, generally accepted to be more pollution-tolerant forms, dominated the periphyton in urban reaches of Coyote Creek.

These preliminary biological investigations in Coyote Creek have indicated distinct differences in the taxonomic composition and relative abundance of the aquatic biota present in various reaches of the stream. The non-urban sections of the creek have been found to support a comparatively diverse assemblage of aquatic organisms, including 12 species of fish and various benthic macroinvertebrate taxa such as mayflies, caddisflies, aquatic beetles, midges, blackflies, snails, and fingernail clams. In contrast, however, the urban portions of the creek have been shown to comprise an aquatic community that is generally lacking in diversity and is dominated by pollution-tolerant organisms such as mosquitofish and tubificid worms.

Table 19. TAXONOMIC COMPOSITION AND RELATIVE ABUNDANCE OF DIATOMS COLLECTED ON GLASS SLIDES IN COYOTE CREEK DURING THE SPRING OF 1978

	Relative Abundance (%) of Each Taxon Within the Sample					
	Non-Urban Reach			Urban Reach		
Taxon	Station 1	Station 2	Station 3	Station 4	Station 5	Station 6
Centrales						
Coscinodiscaceae						
Melosira spp.	0.4	-	-	-	1.2	0.8
Pennales						
Diatomaceae						
Diatoma vulgare	0.4	-	1.5	-	-	-
Fragilariaceae						
Synedra sp.	-	-	-	0.8	0.9	0.4
Achnanthaceae						
Achnanthes lanceolata	20.6	37.8	56.1	49.8	0.9	1.6
Rhoicosphenia curvata	0.4	-	-	1.2	-	-
Cocconeis pediculus	15.0	18.2	0.4	-	-	-
Cocconeis placentula	62.4	44.0	41.2	-	-	-
Naviculaceae						
Navicula spp.	-	-	-	-	10.5	23.8
Diploneis sp.	-	-	-	-	2.4	-
Frustulia rhomboides	-	-	-	-	0.4	-
Gyrosigma sp.	-	-	-	-	-	0.4
Gomphonemataceae						
Gomphonema spp.	-	-	-	2.8	6.9	0.8
Cymbellaceae						
Cymbella spp.	0.8	-	-	-	2.0	0.4
Rhopalodia spp.	-	-	-	-	-	0.4
Nitzschiaceae						
Nitzschia spp.	-	-	0.8	43.4	67.5	70.6
Denticula elegans	-	-	-	-	2.4	0.4
Surirellaceae						
Cymatopleura solea	-	-	-	-	0.9	-
Surirella spp.	-	-	-	2.0	4.0	0.4
Total Number of Frustules/mm^2	5545	4950	1874	4488	1189	4575

Source: Pitt and Bozeman 1979

REFERENCES

Calhoun, A.C., Ed. 1966. Inland Fisheries Management, California Department of Fish and Game. Sacramento, California. 546 pp.

Farmer, J.G., and T.D.B. Lyon. 1977. "Leads in Glasgow Street Dirt and Soil." The Science of the Total Environment, 8:89-93.

McKee, J., and H.W. Wolf. 1963. Water Quality Criteria, 2nd ed.: State Water Quality Control Board. Sacramento, California.

McMullen, T.B., and R.B. Faoro. 1977. "Occurrence of Eleven Metals in Airborne Particulates and Surficial Materials." Journal Air Pollution Control Association. 27:12:1198-1202. December.

Metcalf and Eddy Engineers. 1978. "Surface Water Management Plan for Santa Clara County - Technical Appendices" for Santa Clara Valley Water District. San Jose, California. December.

Moyle, P.B., and R.D. Nichols. 1973. "Ecology of Some Native and Introduced Fishes of the Sierra Nevada Foothills in Central California." Copeia, 1973(3):478-490.

Moyle, P.B. 1976. Inland Fishes of California. University of California Press. Berkeley, California. 405 pp.

Olson, K.W., and R.K. Skogerboe. 1975. "Identification of Soil Lead Compounds from Automotive Sources." Environmental Science and Technology, 9:3:227-230. March.

Pitt, R.E., and G. Amy. 1973. Toxic Materials Analyses of Street Surface Contaminants. EPA-R2-73-283. U.S. Environmental Protection Agency. Washington, D.C. August.

Pitt, R.E. 1979. Demonstration of Non-Point Pollution Abatement through Improved Street Cleaning Practices. EPA Grant No. S-804432.

Pitt, R.E., and M. Bozeman. 1979. Water Quality and Biological Effects of Urban Runoff on Coyote Creek - First Phase. EPA Grant No. R805418010.

U.S. Environmental Protection Agency. (undated). 1975 Interim Primary Drinking Water Standards: Subchapter D, Part 141, Subpart A.

_____. 1973. Proposed Criteria for Water Quality: Vol. 1. October.

_____. 1973. Water Quality Criteria, Environmental Studies Board: NAS-NAE, EPA-R3-73-033. March.

8

Seventh Session

METHODOLOGIES FOR STORMWATER IMPACTS ASSESSMENT

Moderator: Wayne C. Huber
University of Florida
Gainesville, Florida

NATIONWIDE ASSESSMENT OF URBAN STORMWATER POLLUTION IMPACTS ON RECEIVING WATER BODIES

by

James P. Heaney, Wayne C. Huber,
and Melvin E. Lehman

University of Florida, Gainesville, Florida 32611

ABSTRACT

Urban stormwater runoff has been recognized in recent years as a potential major contributor of pollution to receiving water bodies. Assessment of urban stormwater runoff pollutant quantities and characteristics have been made for several areas throughout the United States, the most ambitious being the Environmental Protection Agency's 208 Areawide Wastewater Management Planning Program. Price tags for abating urban stormwater pollution (through elimination or reduction of discharges) range in the billions of dollars. Projections of high costs have forced a look beyond abatement of discharges to the receiving water bodies for insight as to what are the impacts, where are they and are they significant?

First-year results of a nationwide search for documented case studies of impacts of urban runoff receiving waters indicate that well-documented cases are scarce. Impacts previously attributed to urban stormwater runoff may be point source impacts in disguise, or they may be masked by greater contributions from other sources. In some cases they are offset by hydrological, biological, or geological attributes of the receiving water body.

The lack of documentation and clear definition of urban stormwater impacts makes the task of assessing importance of this pollution source even more difficult. Efforts to address this aspect include relating sources of pollutants and pollutant types to receiving water characteristics and effects on desired water uses. Characteristics such as stream or lake bed hydraulics, present and potential water uses, established stream standards, ecological data and water quality information are being summarized for the documented cases to determine how the urban runoff pollutants might behave or react in the receiving water and what potential use they might affect most adversely. Results of these analyses are to be used as a basis for devising simple criteria for analyzing an urban area to determine whether a potential impact does or would occur.

INTRODUCTION

A previous nationwide assessment indicated that urban runoff can be viewed as causing problems since, on a nationwide average, the quantity of urban runoff (13.4 in/yr) is approximately equal to the quantity of sewage (12.8 in/yr), and the annual BOD_5 per acre from a sewage treatment plant with a removal efficiency of 90 percent is 59.4 pounds as compared to 43.6 pounds from urban runoff (1). Furthermore, the cost of controlling urban runoff appears to be competitive with the cost of additional removal of pollutants in sewage. Consequently, if further reductions in pollutant loads are needed, then urban runoff controls as well as further waste treatment should be evaluated carefully. The anticipated high price tag for such control programs has prompted decision makers to take a harder look at how serious the problem really is. This paper presents the results of a year-long effort to search published and unpublished literature, 201 and 208 project documents, EPA-furnished project materials, agency data and permit files, and other miscellaneous data sources to characterize stormwater runoff impacts on receiving waters.

IMPACTS DEFINED

Several interrelated views on impact assessment may be gleaned from a review of the literature. Traditionally, two perspectives, public health and sanitary engineering, were of prime importance. The public health approach focused on prevention, whereas sanitary engineers took a cost-effectiveness approach (2). An example from the turn-of-the-century is the controversy over whether cities should be required to treat their waste to reduce downstream water treatment costs. Sanitary engineers argued that the assimilative capacity of the rivers should be considered and that it is much more cost-effective to treat the intake water than to spend larger sums (approximately ten times more) on upstream waste treatment. Cooperative efforts between these two groups led to the present regulatory approach, wherein effluent and/or receiving water standards are promulgated. Within this context, "impacts" can be defined in terms of whether the "standards" have been violated.

This approach prevailed until 1972 when the Federal Water Pollution Control Act Amendments established the following basic water quality goals and policies for the United States (3, 4):

1. The discharge of pollutants into navigable waters should be eliminated by 1985.

2. Wherever attainable, an interim goal of water quality, which provides for the protection and propagation of fish, shellfish, and wildlife and for recreation in and out of water, should be achieved by July 1, 1983.

3. The discharge of pollutants in toxic amounts should be prohibited.

These amendments represented a shift towards the early public health philosophy of anti-degradation with relatively little consideration being given to the cost of attaining these goals.

In this same general period, the National Environmental Policy Act (NEPA) of 1969 ushered in a new field of environmental impact assessment. Numerous methodologies were proposed by professionals from various disciplines. Canter (4) outlines the following 12-step procedure for the water environment.

1. Determine types and quantities of water pollutants emitted from all alternatives for meeting a given need during both construction and operational phases.

2. Determine the existing water quantity and quality levels for the surface watercourses in the area. Examine the frequency distributions and the median and mean data for both water quantity and quality. If possible, consider historical trends of water quality. Note particularly the low flow utilized by the local regulatory agency for maintenance of water quality standards.

3. Document unique pollution problems that have occurred or are existing in local surface watercourses.

4. If relevant for the project alternatives, describe groundwater quantity and quality in the area, noting the depth of the groundwater table and direction of groundwater flow. Identify major local uses of groundwater, and delineate historical trends for groundwater depletion and pollution.

5. Assemble summary of key meteorological parameters for the area, noting particularly the monthly averages of precipitation, evaporation, and temperature.

6. Procure the applicable water quality standards for local surface watercourses and groundwater supplies if relevant. Specify applicability of effluent standards and required treatment technology and state whether the receiving stream is water-quality limited or effluent limited. Consider time schedules required for attaining applicable water quality standards.

7. Summarize the organic waste load allocation study for the area. Also procure extent information on inorganic, thermal, sediment, and bacterial waste loads. Identify known point sources of pollution, focusing specifically on unique discharges or wastewater constituents. Also enumerate the types of water uses in the area and summarize the quantities involved.

8. Determine the mesoscale impacts by calculating estimated daily quantities of water pollutants from the alternatives during both construction and operational phases and comparing these to existing

waste loads in the drainage area. Determine the percentage increase in these waste loads. Note existing water quality parameters that are good or poor relative to current or potential standards.

9. Consider construction phase impacts in terms of the following factors:

   a. Time period of construction and the resultant time period of decreased water quality. Specify stream discharges and quality variations that would be anticipated during the construction phase.

   b. Anticipated distance downstream of decreased water quality.

   c. Implications of decreased water quality relative to downstream water users. If there are users that require certain water qualities, identify the required raw water quality characteristics and discuss the effects of decreased quality during the construction phase.

   d. Specific construction specifications directed toward pollutant minimization.

10. Determine the microscale impacts by calculating specific downstream concentrations resulting from conservative pollutants, dissolved oxygen concentrations resulting from nonconservative (organic) pollutants, and temperatures resulting from thermal discharges. Consider these microscale impacts for both construction and operational phases. Compare calculated downstream concentrations with applicable water quality standards. Check if applicable effluent standards are met for existing facilities, or consider how they will be brought into compliance. In the case of new sources, identify necessary technology for compliance with new source performance standards.

11. If water quality or effluent standards are exceeded, consider mitigation or control measures.

12. Consider operational impacts of alternatives in terms of the following factors:

   a. Frequency distribution of decreased quality and quantity.

   b. Effects of sedimentation on the stream bottom ecosystem.

   c. Fate of nutrients by incorporation into biomass.

   d. Reconcentration of metals, pesticides, or radionuclides into the food web.

e. Chemical precipitation or oxidation/reduction of inorganic chemicals.

f. Anticipated distance downstream of decreased water quality and the implications for water users and related raw water quality requirements.

g. General effects of any water quality changes on the stream ecosystem.

h. Unique water quality changes that occur as a result of water impoundment and thermal stratification.

This method goes well beyond the strict determination of whether standards have been violated. The output from this assessment is sometimes summarized in terms of a numerical quality index [see Dunette (5) or Ott (6) for a summary of the literature on this subject].

Ecologists provided yet another perspective on impact assessment. They called for a more holistic view of the problem and a recognition that measures of value other than man's are relevant. Ehrenfeld (7) summarizes the viewpoints on how "non-resources" can be valued in an appropriate manner. He lists nine categories of "economic" values:

1. recreation and aesthetic values,

2. undiscovered or undeveloped,

3. ecosystem stabilization,

4. examples of survival,

5. environmental baseline and monitoring values,

6. scientific research values,

7. teaching values,

8. habitat reconstruction values, and

9. conservation values: avoidance of irreversible change.

In addition, he lists two "non-economic" values:
1. natural art value, and
2. religious value.

Lastly, economists have contributed a significant body of literature on this subject. Freeman (8) summarizes the categories of impacts according to the channels through which environmental changes affect man.

1. Through living systems--biological mechanisms

    Human health--morbidity, mortality
    Economic productivity of ecological systems--agricultural productivity, forestry, fisheries.

2. Other ecosystem impacts

    Recreational uses of ecosystems--fishing, hunting
    Ecological changes--diversity, stability

3. Through nonliving systems

    Materials damages, soiling, production costs
    Weather, climate
    Other--odor, visibility, visual aesthetics.

To avoid the accusation of parochialism in adopting, _a priori_, any one or combination of the above systems for assessing stormwater "impacts," the literature search was approached with an open mind. However, the need for a more precise definition of an "impact" became apparent early into the literature search. Definitions of "impact" are almost as numerous as there are investigators, congressmen, regulators, and citizen review committees of urban stormwater problems. The range of impact definitions includes specific cause-effect statements, comparisons of constituent concentrations to numerical standards, sensory perceptions such as odor and color problems, and "perceived" impacts from citizens. All are applicable and valued, with respect to the level of action or understanding desired. However, for a "standard" definition by which to conduct comparative studies at the environmental regulatory agency (EPA) level, a stormwater impact was defined as one which resulted in "loss of beneficial use." Beneficial uses considered are those listed in local, state, and federal water laws, which include drinking water use, fishing and shellfishing, swimming, boating, manufacturing process water use, etc. The list of beneficial uses could be extended to include loss of a receiving stream to wastewater disposal as a result of already overloaded assimilative capabilities or as loss of biological integrity, but these uses are not clearly defined in many water laws.

This definition states that an impact occurs when beneficial use is lost as a result of stormwater runoff. The key question for determining documentation of impacts is _at what point does loss of beneficial use actually occur_? Does loss occur when a receiving water body does not meet a governmental water use classification, when constituent levels violate water quality standards (or criteria), when 30-year population counts show reductions in fish populations, or when people get sick or fish kills occur? When is loss of use implied, when is it imminent, and when is it obvious? The following summary relates "impact level" to "loss of use" in a general overview:

Impact Level	Loss of Use
Policy or Management Planning	Is considered possible or is implied.
Standards or Criteria Violations	Is implied, may be imminent, or can actually occur through restriction of use.
Documented Cases of Cause-Effect	Actually occurs.

Policy or Management Planning--At this level the loss of beneficial use is implied but has not actually occurred. Use of a receiving water body for stormwater discharge may violate (or be in contrast to) a comprehensive plan, environmental agency philosophy, coastal zone management policy, area-wide water use classification system, or some other indicator of intended use.

Standards or Criteria Violations--This is the level at which impact typically has been assessed. The usual approach is to measure constituents of storm or receiving waters, compare measured values with local, state, or federal standards (criteria, or guidelines), and then directly equate impact with the number of constituent standards violated, or the number of days a constituent standard is violated. This same approach is used in cases where key concentrations have not been identified or developed. The presence or absence (occurrence) of a constituent (e.g., EPA list of 129 priority pollutants) is often considered an impact.

The loss of use at this level can be implied, may be imminent, or can actually occur. Whether or not an impact actually occurs is relative to the basis on which a standard is promulgated. In the absence of supporting data, standards are usually set conservatively, so many documented violations of standards *imply* impact, rather than actually *indicate* impact (e.g., oxygen standards may be violated, but fish kills may not occur). Standards or criteria violations may be considered "paper" impacts.

Documented Cases of Cause-Effect--This was considered the impact level where loss of beneficial use was actually documented. Site-specific evidence of fish kills, beach closings, loss of water supply, trends in aquatic systems deterioration, citizens' complaints of odor, floating debris, medical records of water-use related diseases, and other effects (impacts) caused by or related to stormwater runoff were reviewed. The nationwide survey focused on this level.

Documentation of extreme or unusual events is often easier than identification of trends or subtle changes, especially where complex systems of man and nature are concerned. Questions arise as to the cause of an impact (such as a fish kill). Was it due to an event such as a toxic substance being flushed, or was it due to bioaccumulation (to lethal levels) of a substance over 30 years? Thus, the cause of an impact may be an event, or a series of events establishing a trend. In general, pulsed (event-driven) systems are under examination when addressing stormwater issues, so causes of impacts considered were short-term. The impact itself, being event- or trend-induced can be manifested quickly or over a long period, and finally, the manifested impact may have short or long-term significance. This study attempted to categorize impacts in terms of:

Event-type - short-term impacts such as fish kills and temporary beach closings.

Trend type - long-term impacts such as evidence of biological deterioration or loss of species diversity.

SEARCH FOR IMPACTS

Literature Riview

A systematic category-by-category search and review of literature sources was conducted. Major sources include:

1. EPA Cincinnati in-house files on impacts of urban runoff (9).

2. Review of 208 studies throughout the United States. As part of EPA's National Urban Runoff Program (NURP), EPA Headquarters personnel and a team of consultants visited every regional office at least once. During the initial visit, the regional personnel were asked to identify which of these 208 studies indicated that urban runoff was a "problem." Next, the group was asked which areas had receiving water data documenting that an impact existed. They were asked to indicate whether the "problem" related to violation of water quality standards, impairment of beneficial use(s), aesthetics, or other cause(s). Lastly, each regional group was asked to suggest candidate cities for further study. Ideally, these cities should have a clearly identified urban runoff problem, and sufficient interest in solving it to finance 25 percent of the cost of the study. Based on this procedure, 30 cities were selected for further study.

3. Computerized Literature Searches

    a. Water Resources Scientific Information Catalog (WRSIC) system of U.S. Department of Interior, Office of Water Research and Technology.

    b. Smithsonian Scientific Information Exchange (SSIE) lists of on-going research projects.

    c. University of Florida's State Technologies Application Center (STAC) information retrieval system which is tied into the NASA system of about 20 million publications.

4. U.S. Environmental Protection Agency Fish-Kill Data. Approximately 10,000 individual fish-kill reports were surveyed [see (10) for a general summary].

5. Studies for the National Commission on Water Quality on beach closings (11).

6. 1978 Needs Survey (12).

7. EPA National Urban Runoff Program files of case studies of 30 cities.

8. Sutron Corp. case studies on relationships of rainfall and dissolved oxygen (13).

9. Abstracts for EPA National Conference titled Urban Stormwater and Combined Sewer Overflow--Impact on Receiving Waters, November 1979.

10. Environmental Protection Agency, Nationwide 201 and 208 Technical Documents and Wastewater Management Plans.

11. National Eutrophication Survey Documents (14).

12. 1974 EPA National Water Quality Inventory (15).

The literature survey was the main tool used to identify case study areas. Other sources included review of newspaper articles, attendance at scientific and technical conferences, telephone conferences with regional Environmental Protection Agency offices, and conversations with other investigators in the field.

Handling of Receiving Water Information

The documents from the preliminary screening as a result of the literature review were catalogued and incorporated into the existing EPA Urban Stormwater Runoff Data Base geographical files at the University of Florida. The information is summarized, by state, for each of the 248 urbanized areas listed in (1). Case studies for smaller cities are included in the general information regarding each state. For each urbanized area, the following information is presented:

Demographic Data:

--1970 population and urbanized area acreage from (1)
--Percent combined sewers from (12)

Hydrologic Background:

--General information regarding the water resources of the area. Most of this information was taken from a report by Schneider (16).

Waste Sources:

--The annual volume of sewage and urban runoff from the area. The data are taken from (1).

Receiving Waters:

--The primary receiving water is indicated. Minor receiving waters or special study sites are also mentioned. Where appropriate, a dilution ratio is calculated as follows:

$$\text{Dilution Ratio} = \frac{\text{Primary Receiving Water Flow (in/yr)}}{\text{Sewage flow (in/yr)} + \text{Urban Runoff (in/yr)}}$$

All values are in inches per year over the developed urbanized area.

Specific Studies:

--Summaries of receiving water impact studies.

Maps:

--For each city, a 1:500,000 map was drawn from the U.S.G.S. State Hydrologic Maps. The same scale was used for all cities. Hydrologic boundaries are indicated. Receiving waters are shown exactly as indicated on the original map. The name of the receiving water is included only if it was included on the original map. This provides the user with a better perspective on the relative importance of the local receiving waters.

Latitude and longitude are included. The location of significant U.S.G.S. sampling stations is shown.

Streamflow Data:

--The 1950-1960 summary of U.S.G.S. monthly flow data are included. This summary includes the long-term average flow and extreme values.

A sample result for Tampa, Florida is shown in Appendix A.

RESULTS

Types of Receiving Waters

Table 1 shows the percentage of urbanized areas that discharge into each of 15 categories of receiving waters. The results indicate that about 75 percent of the discharge is to rivers, 5 percent to lakes, and 20 percent to estuaries or oceans. However, this distribution can be misleading because many of the impacted areas of major significance are not the primary receiving water.

TABLE 1. DISTRIBUTION OF PRIMARY RECEIVING WATERS FOR URBANIZED AREAS IN THE UNITED STATES

Category	Percent of Total
A. Rivers	
1. Creeks and shallow streams [depth (d) <2 feet]	8.8
2. Upstream feeders ($2 \leq d < 5$)	13.3
3. Intermediate channels ($5 \leq d < 10$)	15.1
4. Main drainage rivers ($10 \leq d < 20$)	21.0
5. Large rivers ($20 \leq d < 30$)	12.3
6. Impounded rivers (flow controlled or depth ≥ 30 feet)	4.5
Sub-Total, Rivers	75.0
B. Lakes	
1. Small ponds, back waters	0.7
2. Lakes	4.2
Sub-Total, Lakes	4.9
C. Estuaries and Ocean	
1. Shallow estuary or bay ($d < 10$ feet)	3.5
2. Medium depth estuary or bay ($10 \leq d < 30$)	5.3
3. Deep estuary or bay ($d > 30$)	3.9
4. Open ocean or beach	7.4
Sub-Total, Estuaries and Oceans	20.1
Total, A,B, and C	100.0

*Source: Raw Data From 1978 NEEDS Survey (12).

Documentation of Impacts

Fish Kills--

Urban receiving water bodies, for a variety of reasons other than stormwater runoff (e.g., habitat alteration, multi-source pollutants, temperature), may not have abundant fish populations. In fact, abundance of aquatic life in general is most often inversely proportional to the degree of urbanization. However, many fish kills have been related to stormwater runoff, combined sewer overflows (CSO's), storm sewers, or rainfall events. Table 2 is a summary of storm-water/storm-sewer related fish kill reports in the EPA files (17) for the period from January 1970 to May 1979. It gives the number of fish kills reported for a given year relative to the seven categories listed. In order of appearance in the table, a brief description of each category is presented.

Pollutant Spilled into Storm Sewer--The pollutant in this category may or may not have been flushed by a rainfall event (not indicated on fish kill report). Typically this is a human event-type impact where someone dumps a toxic substance into a storm drain, or a toxic substance accidentally spills and drains into a storm sewer.

Storm Sewer Discharge from Rainfall Event--This is distinguished from the previous category in that the person filling out the fish-kill report stated that the kill was due to discharge during a storm event. This category overlaps the previous one in instances where residuals of a spilled toxic substance remain in the storm sewer until adequate flushing occurs.

Combined Sewer Outfall--Fish kills occur downstream of identified combined sewer outfalls (CSO's), but are not necessarily storm-event related.

Rainfall-Runoff Related to Land Use--This fish kill category includes reports which do not specify drainage ways, storm sewers, or sheet flow but document fish kills following rainfall-runoff events. The attempt to classify relative to land use is a very rough estimate. Fish-kill reports are not detailed enough to identify the specific sources of runoff, or how the runoff actually gets into the receiving water body.

Acid Mine Drainage-Storm Event Related--Acid mine drainage, a nonurban stormwater problem, was included in the table for two reasons: the mining activity may be borrow material for roads, or a town built around a mine; and it is useful to compare the number of acid mine related fish kills to those related to stormwater/storm sewers. Acid mine drainage is a fairly well documented problem; the numbers of reported kills are comparable.

Landfill Leachate Storm-Event Related--Most landfills are products of urban activity, even though many are located in agricultural or fringe areas.

Other--This category includes fish-kill reports not suitable under the other headings but of interest due to location (a receiving water body under

TABLE 2. SUMMARY OF NUMBER OF STORM-WATER/STORM-SEWER RELATED FISH-KILL REPORTS, U.S. EPA DATA, 1970 to MAY, 1979 BY CAUSE OF KILL*

Year	Pollutant Spilled into Storm sewer	Storm Sewer Discharge From Rainfall Event	Combined Sewer Outfall	Rainfall-Runoff Related to Land Use			Acid Mine Drainage-Storm Event Related	Landfill Leachate-Storm Event Related	Other
				Comm/Ind	Residential/ Suburban	Agric.			
1970	7	4	3	3	7	4	3	2	3
1971	4	2	3	3	11	12	2	2	7
1972	1	2	3	1	4	9	5	8	1
1973	-	2	1	3	7	4	3	5	4
1974	6	3	1	3	5	15	2	1	-
1975	6	-	3	3	5	5	-	1	1
1976	2	2	1	2	6	7	-	1	4
1977	5	2	-	3	3	2	2	1	2
1978	1	2	-	-	1	5	-	-	1
1979	-	1	-	1	-	1	-	2	-
Total	32	20	15	22	49	64	17	23	23

Comm = commercial land use
Ind = industrial land use
Agric = agricultural land use

*Source: Raw Data from EPA (17)

stormwater runoff study, e.g. Trinity River, Texas), or apparent storm-water-related kill (indicated, not directly stated).

Some general comments pertinent to the survey of the fish-kill data are:

1. Because many urban streams are grossly polluted from a variety of sources, no fish remain. Hence, no fish kills are reported.

2. Very few of the reports (a total of 20 for the period 1970-1979) state storm-water runoff directly as the cause of fish kills.

3. There are several instances of stormwater flushing pesticides, sewage deposits, herbicides, dumps of oil, etc.

4. There are several instances of accidental dumps into storm sewers of toxic substances and then flushed prematurely of rainfall events--a good example is a hotel or drug store fire where fire department runoff goes into a storm drain.

5. Many citations indicate fish kills due to "eutrophication," or "natural causes." These were not included in Table 2. General water quality deterioration or nutrient enrichment, could in part, be due to storm water runoff.

6. The EPA instructions for filling out fish-kill reports group sewerage, storm water, and CSO's into a single category (sewerage system). If the person filling out the form does not specify the type of system, it is anyones "best guess" as to which type of system caused the kill. The Sewerage System category contains the largest number of fish kills (10). Unless CSO's or storm sewers were specifically cited, they were not counted and recorded in Table 2. For this reason, Table 2 probably understates the actual number of CSO or storm sewer-related kills.

7. In a true urban stream, the resident fish population may be adapted to pollutant loads, and/or pulses; or the population may become dominated by more pollutant tolerant species. The fish-kill data do have some information concerning species of fish killed, but no historical information is available to assess impacts of adaptive or community changes.

8. Fish-kill frequency data are relative. If an area experiences a severe kill that wipes out the fish population, subsequent events that occur do not record kills. Conversely, a severe kill could wipe out a resident population, but immigration from a nearby water body would mask reduction in the local population on a longer term basis.

9. Fish-kill reports are prepared by people with a variety of positions and backgrounds. The inherent variability in such a nationwide

reporting system makes numbers and statistics for this type of data base very subjective and less reliable.

The monthly distribution of fish kills as a percentage of the total number of fish killed and the total number of reports is shown in Figure 1. As expected, relatively more kills occur during the warmer months of the year.

Beach Closings--

The National Commission on Water Quality placed heavy emphasis on attempting to evaluate the benefits associated with water pollution control (11). A total of 3,521 beaches throughout the United States were surveyed. Of these beaches, 449 had water quality problems. Table 3 shows the proportion of the closings due to various causes. While urban runoff is not identified as a separate category, the majority of the closings may be attributable to this cause. For example, within the coliform related problem category, almost 50 percent of the total closings are due to undefined sewage contamination or unknown causes.

Low Dissolved Oxygen--

Keefer, Simons, and McQuivey (13), in an EPA sponsored national assessment, related the magnitude of dissolved oxygen (D.O.) deficits and the presence of storm runoff downstream of urban areas. Based on an initial screening of over 1,000 D.O. monitors located throughout the United States, over 100 water quality monitoring sites in and downstream of urban areas were selected. Approximately one-third of these monitors indicated at least a 60 percent probability of a higher than average dissolved oxygen deficit occurring at times of higher than average streamflow. Of the areas where a low D.O., high streamflow correlation was observed, a more detailed hourly analysis was performed. These results were striking. During steady state low flow conditions the D.O. fluctuates diurnally between 1 and 7 mg/l. However, after a storm begins, the diurnal fluctuations are completely dampened. The minimum wet-weather D.O. is 1 to 1.5 mg/l lower than the dry-weather minimum, and remains that way for 1 to 5 days. As the impact of the storm dissipates, the D.O. resumes its original cyclic behavior. This relationship, for the Scioto River at Chillicothe, Ohio, (downstream of Columbus, Ohio) is shown in Figure 2. It suggests the need to reexamine the traditional approach to defining "critical" conditions in receiving waters.

In this study (13), the two most severe cases of low D.O. were the Trinity River near Dallas, Texas and Wilsons Creek near Springfield, Missouri. The authors' independent analysis indicates that both of these receiving waters have large deposits of sludge from primary sewage treatment plants. Thus, a significant part of the problem is attributable to resuspension of this benthal material.

National Urban Runoff Program Case Studies--

This detailed evaluation of urban runoff problems in cities throughout the Unites States yielded 30 case studies. Each of these applicants

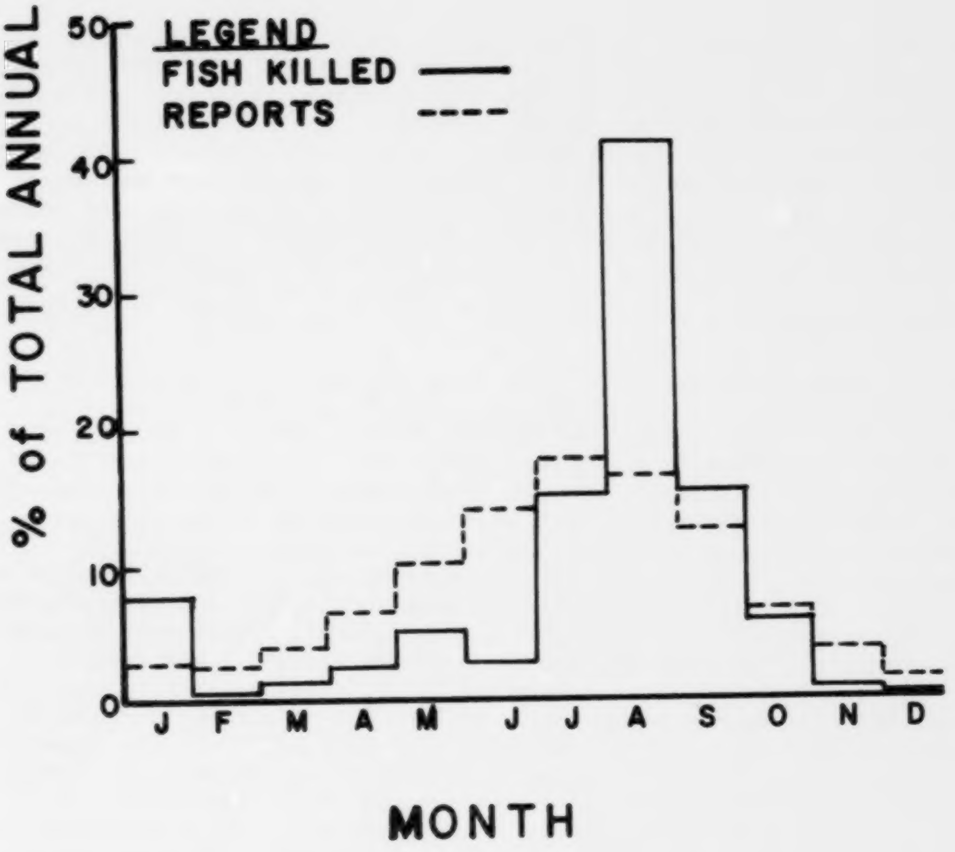


FIGURE 1. MONTHLY DISTRIBUTION OF FISH KILLS AS A PERCENTAGE OF TOTAL FISH KILLED

TABLE 3. CAUSES OF WATER QUALITY RELATED BEACH CLOSINGS IN THE UNITED STATES

Cause	Number	Percent of Total
Algae, Scum	11	1.8
Turbidity	13	2.2
High Coliform or Fecal Coliform Count Due To:		
Flood, Wind, Heavy Rainfall	41	6.8
Agricultural Runoff	10	1.7
Sewage Treatment Plant Malfunction, Spills, Outfall	42	6.9
Undefined Sewage Contamination or Unknown	337	55.7
Other	71	11.7
Unknown	80	13.2
Total	605	100.0

*Source: Raw Data From Battelle (11).

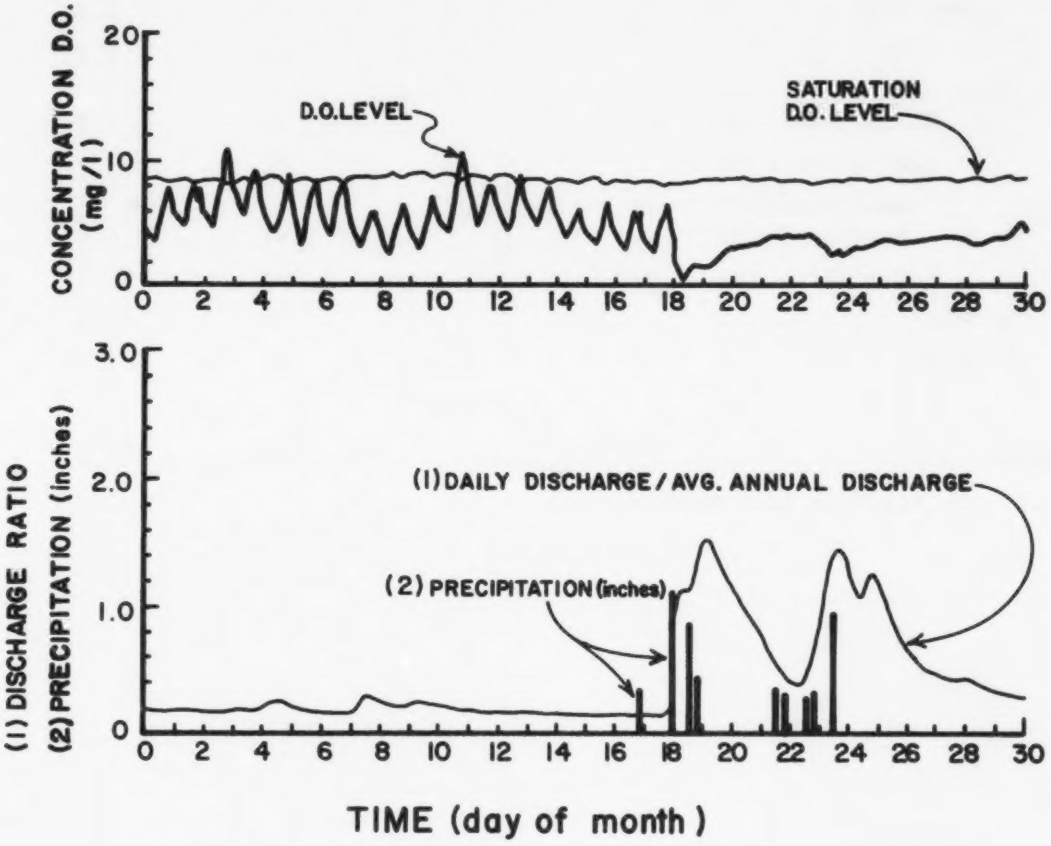


FIGURE 2. OBSERVED RELATIONSHIP BETWEEN DIURNAL DISSOLVED OXYGEN CONCENTRATION AND STORM EVENTS FOR THE SCIOTO RIVER AT CHILLICOTHE, OHIO

indicated that they have a "problem" and that they were willing to do something about it. However, little definitive evidence was presented to support the contention of a receiving water problem. Summary information regarding these projects is shown in Table 4. Descriptions of the receiving water problem for some of these projects are presented below for illustrative purposes.

1. Baltimore, Maryland--Studies by Olivieri and his co-workers (18) appear to be the best available source of information on the bacteriological quality of urban stormwater. Their results indicate that "urban runoff" is really a composite of all unaccounted for residuals leaving an urban area via the watercourses. It includes illicit industrial waste, cross connections with the sanitary sewer system, septic tank seepage, landfill leachate, etc. Thus, sanitary surveys of local receiving waters are needed to characterize the actual problems that exist.

2. Myrtle Beach, South Carolina--Bacteriological contamination of the city's beaches occurs after heavy rains. Urban runoff is the alleged cause although it might be illicit interconnections with the sanitary sewer system.

3. Tampa, Florida--Deterioration of the City of Tampa surface water supply in the Upper Hillsborough River appears to be partially attributable to urbanization of the riparian lands. Water quality in the Lower Hillsborough River and Hillsborough Bay is degraded by landfill leachate, contaminated spring water, sanitary sewer system overflows, zoo runoff, as well as the more general forms of urban runoff. These receiving waters have been seriously impacted by sewage and water treatment plant sludges, industrial wastes, incinerator leachate, and other problems. The relative importance of urban runoff is to be determined.

4. Milwaukee, Wisconsin--Milwaukee has very serious combined sewer overflow problems and a large accumulation of sewage sludges. The problem is relatively well documented and the City of Milwaukee is committed to take remedial actions.

5. Austin, Texas--Town Lake, a water supply source for the city, has received urban runoff for a number of years. Urbanization is proceeding upstream along the lake. There is general evidence that water quality in the lake has deteriorated but the exact extent is unknown. Are stringent controls on urban runoff needed and/or should the City of Austin move its water supply further upstream?

6. Bellevue, Washington--The City of Bellevue seeks to preserve the salmon runs. Efforts are being made to prevent deterioration of the local streams. This requires effective ordinances and extensive monitoring. What would such a program cost and how effective will it be?

TABLE 4
CHARACTERISTICS OF NATIONWIDE URBAN RUNOFF PROGRAM PROTOTYPE PROJECTS

			Receiving Water Type						Impacts			Beneficial Uses				
Region	Applicant Agency	Project Location	Small Stream	Lake	River	Estuary	Ocean	Groundwater	Actual Impairment of Beneficial Use	Standards Violation	Public Concern	Water Supply	Contact Recreation	Non-Contact Recreation	Fish and Wildlife	Aesthetics
I	Comm. of Mass	Lake Quinsigamond, MA		●					●		○		●	○	●	○
	Comm. of Mass.	Mystic River, MA	○	●	○				●	○			●	○	○	
	N.H. Water Supply	Durham, NH			○	●			●	○	○			○	●	
		Vermont														
II	Long Island RPB	Long Island, NY			●	○		○	●	○	○	●	○	○	●	
	New York DEC	Lake George, NY		●							●	○	●	○		
	New York DEC	Irondequoit Bay, NY		●					●	○	○		●	○	○	
III	Washington COG	Metro Washington, DC	○	○	○	○			●	○	○	●	○	○	○	○
	Baltimore RPC	Baltimore, MD	○	○	●				○	●	○			○	●	
IV	Waccamaw RPC	Myrtle Beach, SC					●		○	●	○		●			
	N. Carolina DR	Winston-Salem, NC	○	○	●					●						
	Tampa DPW	Tampa, FL		○	○	○			○	●		○	○	○	○	○
V	Tri-County RPC	Lansing, MI			○					○					○	
	SEMCOG	Oakland County, MI		○							○			○		○
	SEMCOG	Ann Arbor, MI			●						○	○	○	○		
	IEPA	Champaign-Urbana, IL	○							●					○	○
	NIPC	Chicago, IL	○	○							○		○		○	○
	Wisc. DNR/SEWRPC	Milwaukee, WI	●	○	○					●			○	○	○	
VI	City of Austin	Austin, TX		●							○	○	○	○		○
	Metroplan	Little Rock, AR	○		●					○	○			○	○	
		Albuquerque, NM														
VII		Kansas City, MO														
VIII	Denver RCOG	Denver, CO	○	○	●				○	○		○	○	○	○	
	Salt Lake County	Salt Lake City, UT	○		●					●		○		●	○	
	6th District COG	Rapid City, SD														
IX	Alameda CFC & WCD	Castro Valley, CA	●													
	SCAG	Santa Ana, CA				●			●		○		○	○	●	○
		Fresno, CA														
X	City of Bellevue	Bellevue, WA	●	○					●		○				●	○
	Lane County COG	Eugene, OR	○	○	○				○	○			○	○	○	
	Portland MSD	Portland, OR	○							○				○	○	○

LEGEND: ● Aspects of program receiving major emphasis. For receiving waters, this means an emphasis on monitoring and impact assessment. For BMP's, this means an emphasis on determining control effectiveness and cost.

○ Aspects of program receiving less vigorous attention.

Reproduced from
best available copy.

TABLE 4
CHARACTERISTICS OF NATIONWIDE URBAN RUNOFF PROGRAM PROTOTYPE PROJECTS
(CONTINUED, PAGE 2 OF 2)

			Problem Pollutants								Candidate Best Management Practices					
Region	Applicant Agency	Project Location	BOD, COD, DO	Nutrients	Sediments	Heavy Metals	Other Toxics	Oil and Grease	Coliform Bacteria	Average Rainfall (in/yr)	Street Sweeping	Detention Basins	Catch-Basin Cleaning	Cross-Connection Elimination	Wetlands Treatment	Other
I	Comm. of Mass	Lake Quinsigamond, MA		●	○	○			●	66						
	Comm. of Mass.	Mystic River, MA	○	○		○			●							swirl conc.
	N.H. Water Supply	Durham, NH	○	○	○				●		○	○	○			
		Vermont														
II	Long Island RPB	Long Island, NY		○			○		●	44	○	●			●	perm. sewer
	New York DEC	Lake George, NY		●	○	○			○	35						
	New York DEC	Irondequoit Bay, NY	○	●	○	○			○	32						
III	Washington COG	Metro Washington, DC	○	●	○	○			○	40		●				
	Baltimore RPC	Baltimore, MD	○		○	○	○	○	○	45						
IV	Waccamaw RPC	Myrtle Beach, SC							●					○		collector/outfall
	N. Carolina DR	Winston-Salem, NC	○	○		○			○	44						
	Tampa DPW	Tampa, FL	○	○					○	49						
V	Tri-County RPC	Lansing, MI			○		○					●				oversize sewer
	SEMCOG	Oakland County, MI		○	○							●				runoff ordinance
	SEMCOG	Ann Arbor, MI		○	○	○						●			●	
	IEPA	Champaign-Urbana, IL				●					●					
	NIPC	Chicago, IL	○	●	○	○			○	34		●				
	Wisc. DNR/SEWRPC	Milwaukee, WI	○	○	○	○				31	●					
VI	City of Austin	Austin, TX		○	○	○						○				runoff ordinance
	Metroplan	Little Rock, AR	○	○		○			○	50						
		Albuquerque, NM														
VII		Kansas City, MO														
VIII	Denver RCOG	Denver, CO		○	○	●			○	16	○	●				runoff ordinance
	Salt Lake County	Salt Lake City, UT	○	○	○			○	○	15		●	○		●	canal storage, education
	6th District COG	Rapid City, SD		○	○	○	○		○	22	●					
IX	Alameda CFC & WCD	Castro Valley, CA														
	SCAG	Santa Ana, CA			○	○		○	○	13	○					oil/grease traps
		Fresno, CA														
X	City of Bellevue	Bellevue, WA	○	○	○	○	○	○	○		●	●				
	Lane County COG	Eugene, OR	○		○	○	○	○							○	
	Portland MSD	Portland, OR			○		○	○	○							

For some projects, the program to assess BMP's will not be formulated until the completion of the first year's problem assessment effort.

TABLE 5. MAJOR WATERWAY RANKINGS-PERCENT PARAMETERS EXCEEDING REFERENCE LEVELS*

0 to 7	7 to 17	Over 17
Upper Missouri River	Rio Grande River	Hudson River
Columbia River	Alabama-Coosa Rivers	Delaware River
Lower Tennessee River	Upper Ohio River	Middle Mississippi River
Snake River	Susquehanna River	Lower Arkansas River
Willamette River	Upper Red River	Lower Ohio River
Boston Harbor	Lower Colorado River	Lower Mississippi River
Upper Mississippi River	Potomac River	Middle Ohio River
Yukon River	Detroit Area-Tributaries	Lower Missouri River
Chicago Area-Lake Michigan	Sacramento River	Chicago Area-Tributaries
Upper Tennessee River	Lower Red River	Mississippi River near Minneapolis
Detroit Area-River	Brazos River	Upper Arkansas River
	Upper Colorado River	Middle Missouri River

*Based on the number of parameters having medians which exceed reference levels selected for comparative purposes.

Source: Data from EPA (15)

1. Under the anti-degradation philosophy espoused by PL 92-500 in 1972, there was less need to devote resources to receiving water impact assessment. Urban runoff did not become recognized as a problem until after 1972. Thus, little attention was given to this problem.

2. Impacts of sewage effluent, industrial wastes, and other discharges mask the impacts of urban runoff. Even when other sources have been reduced or eliminated, their residual impacts in terms of benthal deposits are often still evident.

3. The increased reliance on mathematical models for assessing receiving water impacts reduced the level of effort in field sampling programs.

4. The greatly enhanced emphasis on broad-based environmental impact assessments diverted effort from the more traditional sanitary survey approach to assessing impacts. These studies produced relatively little hard information on impacts from urban runoff.

5. The cost of sampling programs is relatively high due to the intermittent nature of storm events, wide variations in flow and concentration, and general inexperience with this type of activity.

6. Expected impacts from urban runoff are relatively subtle and do not cause obvious large scale problems. Thus, more refined and longer-term sampling efforts are needed to develop reliable cause-effect information. Indeed, if experience in the related area of sediment transport in receiving waters is any indication, it may be many years before these cause-effect relationships are understood.

Most cases of documented impacts were made for minor receiving water bodies. Larger water bodies tended to have sufficient dilution, adequate flushing, or multiple source inputs which obscure impact differentiation. The complexity of the stormwater/receiving water systems is such that general correlations were not possible between characteristics of urban areas and receiving water impacts. Cause-effect comparisons of data show that stormwater impacts at any given site may contradict those at another. It was determined that a case-study approach was probably best for future work and that detailed site-specific data collection is needed to adequately document stormwater runoff impacts on receiving water bodies.

Few, if any, trend-type impacts, or causes of impacts can be identified at this time. Ecological deterioration, eutrophication, reduction of quality of life, alterations of trophic state indices, depletion of animal populations, and reductions in diversity are all equated to some degree with stormwater runoff, but are not well correlated to specific losses of beneficial use. Event-type impacts are the easiest to document and will continue to receive greater emphasis. Trend-type documentation will take years to develop even though there are indicators that they exist.

The significance of stormwater impacts on receiving water bodies cannot be assessed at this time. The sparseness of documented cases, the lack of detailed data, and the general focus of stormwater investigations into water quality and system dynamics (and away from actual impacts) do not provide a substantial basis for determinations.

Nationwide attention has not been focused on impact documentation. Thus, the data bases are just now beginning to be extensive enough to address scientific correlation of constituents with respect to impacts, and because some possible sources of impact documentation have not been searched at site-specific levels, the amount of documentation identified is probably much less than what exists. In other words, documentation is only beginning to be found because the search has just begun.

ACKNOWLEDGMENTS

This paper is based on research sponsored by the Municipal and Environmental Research Laboratory, U.S. Environmental Protection Agency. Mr. John English, the Project Officer, provided valuable in-house information on receiving water impacts. Mr. Dennis Athayde, his staff, and consultants provided information on EPA's Nationwide Urban Runoff Program. Dr. Tim Stuart and his staff were very helpful in providing access to EPA's fish-kill data. Mr. Richard Field and Mr. Doug Ammon kept us current on the activities of EPA's Storm and Combined Sewer Branch in Edison, New Jersey.

REFERENCES

1. Heaney, J.P., et al. Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges, Vol. II: Cost Assessment and Impacts. EPA-600/2-77-064, U.S. Environmental Protection Agency, Cincinnati, Ohio, 1977. 364 pp.

2. Tarr, J.A., and McMichael, F.C. Historic Turning Points in Municipal Water Supply and Wastewater Disposal, 1850-1932. Civil Engineering, Vol. 47, No. 10: 82-86, 1977.

3. Federal Water Pollution Control Act Amendments of 1972. PL 92-500, 92d Congress, S. 2770, October 18, 1972.

4. Canter, L.W. Environmental Impact Assessment. McGraw-Hill Book Co., New York, 1977. 331 pp.

5. Dunette, D.A. A Geographically Variable Water Quality Index Used in Oregon. Journal Water Pollution Control Fed., 51: 53-61, 1979.

6. Ott, W.R. Water Quality Indices: A Survey of Indices Used in the United States, EPA-600/4-78-005. U.S. Environmental Protection Agency, Washington, D.C., 1978.

7. Ehrenfeld, D.W. The Conservation of Non-Resources. American Scientist, Vol. 64: 648-656, 1976.

8. Freeman, A.M. The Benefits of Environmental Improvement. The Johns Hopkins Press, Baltimore, Maryland, 1979. 272 pp.

9. English, J.A. In-House Files on Receiving Water Impacts. U.S. Environmental Protection Agency, Cincinnati, Ohio, 1978.

10. Anonymous Fish Kills Caused by Pollution. Fifteen-Year Summary: 1961-1975. EPA-440/4-78-011. U.S. Environmental Protection Agency, Washington, D.C., 1979. 78 pp.

11. Battelle Memorial Institute. Benefits from Water Pollution Abatement-Beach Closings and Reopenings, Final Report to National Commission on Water Quality. NTIS No. PB-251 221, Washington, D.C., 1975. 107 pp. plus appendices.

12. CH2M-Hill. 1978 Needs Survey, Cost Methodology for Control of Combined Sewer Overflow and Stormwater Discharge. EPA 430/9-79-003. U.S. Environmental Protection Agency, Washington, D.C., 1978. 196 pp. plus appendices.

13. Keefer, T.N., Simons, R.K., and McQuivey, R.S. Dissolved Oxygen Impact from Urban Storm Runoff. EPA Report (Draft), Cincinnati, Ohio, 1978.

14. Seyb, L., and Randolph, K. North American Project. A Study of U.S. Water Bodies. EPA-600/3-77-086. U.S. Environmental Protection Agency, Corvallis, Oregon, 1977. 537 pp.

15. Environmental Protection Agency. National Water Quality Report to the Congress. EPA-44019-74-001 and 002. Vol. 1 and 2. Office of Water Planning and Standards, Washington, D.C., 1974. 305 pp. and 279 pp.

16. Schneider, W.J. Water Data for Metropolitan Areas. U.S. Geological Survey Water Supply Paper 1871. Washington, D.C., 1968. 397 pp.

17. Biernacki, E. In-House Files on Fish Kills. U.S. Environmental Protection Agency, Washington, D.C., 1979.

18. Olivieri, V.P., Kawata, K., and Kruse, C.W. Microorganisms in Urban Stormwater. EPA-600/2-77-087. U.S. Environmental Protection Agency, Edison, New Jersey, 1977. 182 pp.

TAMPA

Demographic data

1970 population - 369,000; Urbanized area - 44,000ac; % combined sewers - 0

Hydrologic background (Schneider, 1968)

The municipal and industrial supplies for the Tampa area come chiefly from the Hillsborough River. Because of seasonal distribution of rainfall and the limited storage capacity of the city reservoir, this source is inadequate during dry periods, and a supplemental supply from a large spring (Sulphur Spring) is utilized. Adequate quantities of water are available from the Floridan aquifer to meet the future water requirements of the area.

Much of the metropolitan area is subject to hurricane damage because of its location near sea level and because of extensive residential development along the waterfront. Tampa, which is in the lower reaches of the Hillsborough River, is subject to flood damage during periods of excessive rainfall. However, this problem will be alleviated in the near future as flood regulation reservoirs and bypass channels are completed upstream from the city. Other problems of major importance are encroachment of saline water on fresh ground-water supplies, disposal of municipal waters, and the effects of the metropolitan complex on the coastal waters.

Precipitation - 52.0 in/yr

Waste sources

Sewage - 11.2 in/yr.; Urban runoff - 19.3 in/yr

Receiving waters

Primary - Hillsborough River
    Mean annual flow - 21.1 in/yr
    Dilution ratio - 0.69
Other - Hillsborough and Tampa Bays

Special studies

The Hillsborough River is the primary river draining the Tampa area. The upstream portion of this river is used as a water supply source for the City of Tampa. It receives urban runoff. The lower portion of the river moves through the City of Tampa where it receives inputs from a variety of sources including landfill leachate, water treatment plant alum sludge, and overloaded sanitary sewers. The discharge from the Hillsborough River enters Hillsborough Bay. The bay has serious water quality problems and extensive sludge accumulations due to the discharge of primary treatment plant effluent until very recently. Tampa is one of the EPA National Urban Runoff Program study areas.

Tampa was one of three cities selected as case studies of estimating the impact of improved water quality on beach closings. (Battelle, 1976). The results are summarized below

Tampa-St. Petersburg, Florida. The population region for this study case included Pasco, Pinellas, Hillsborough, and Manatee Counties with a combined 1970 population of 1,185,664. In addition to the resident population, there are an estimated 4,432,000 businessmen, vacationers, and other travelers coming to this area each year. Many of these individuals came expressly for the purpose of swimming on their vacations. It is estimated that over 80 percent of the tourists participate in the winter and about 60 percent in the summer.

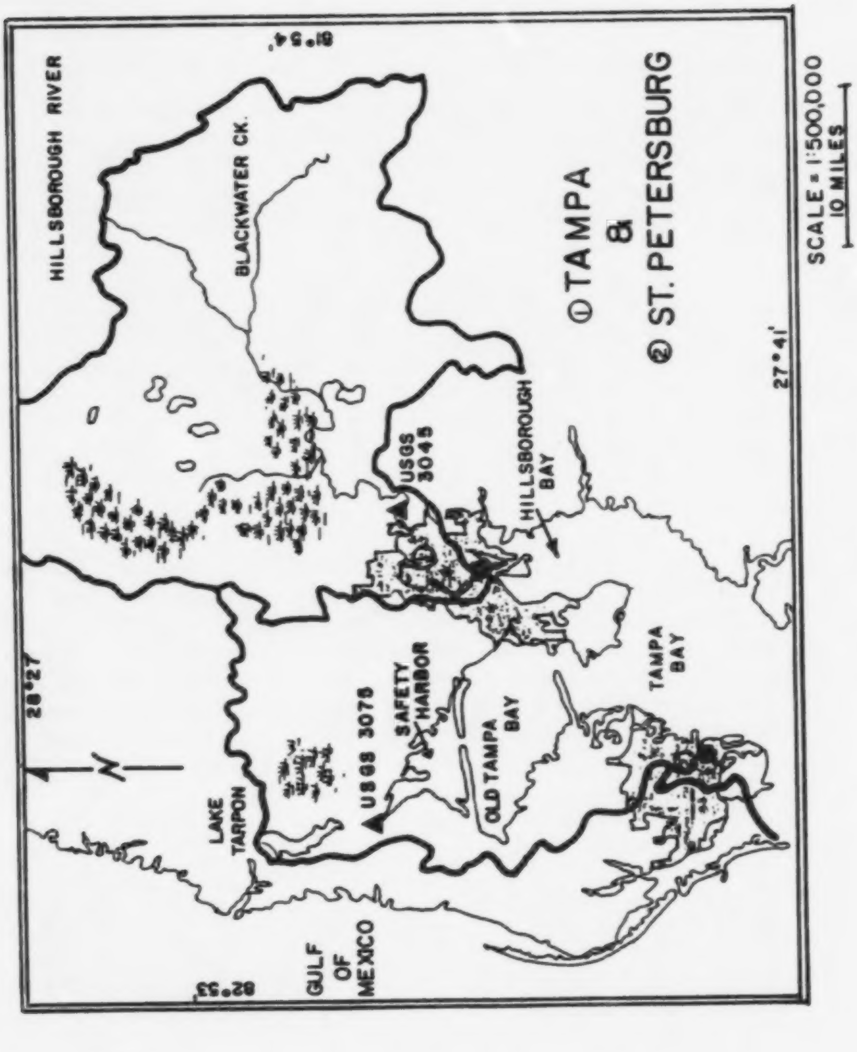
In recent years, six smaller beaches in Pinellas County have been closed intermittently as a result of high coliform counts following heavy rainfall. In Hillsborough County, a beach along the Hillsborough River has been closed permanently since 1972 as a result of bacterial contamination. These areas total 2,450 frontage feet or 1.4 percent of the estimated 178,320 feet of beach frontage in the four-county area.

Because of the availability of abundant high quality beaches, the effect from storm runoff on estimated total resident and tourist swimming activity days is negligible. For the specific beaches that were intermittently affected by high coliform counts, assuming that swimmers avoid them completely, an annual increase of 1 percent in resident and .12 percent tourist activity days were estimated. Because these estimates assume complete seasonal closure, actual loss in activity days would be lower assuming individuals use these beaches intensively when water quality is acceptable. Also, no determination of the specific type and level of use of the affected beaches was obtained during the course of data collection.

References

Federal Water Pollution Control Admin. 1969. Problems and Management of Water Quality in Hillsborough Bay, Florida. NTIS PB-217 147, U.S. Dept. of Commerce, Springfield, Va. 94 pp.

Lopez, M.A., and D.M. Michaelis. 1979. Hydrologic Data from Urban Watersheds in the Tampa Bay Area, Florida, USGS Water-Resources Investigations 78-125. 51 pp.


①TAMPA
&
② ST. PETERSBURG
SCALE = 1:500,000
10 MILES
HILLSBOROUGH RIVER
BLACKWATER CK.
81°54'
27°41'
USGS 3045
HILLSBOROUGH BAY
TAMPA BAY
OLD TAMPA BAY
SAFETY HARBOR
USGS 3075
28°27'
N
LAKE TARPON
GULF OF MEXICO
82°53'

Reproduced from best available copy.

HILLSBOROUGH RIVER BASIN

3045. Hillsborough River near Tampa, Fla.

Location.--Lat 28°01'25", long 82°25'40", in sec.29, T.28 S., R.19 E., on left bank just upstream from spillway of Tampa reservoir dam, at Thirtieth Street, 5½ miles northeast of Tampa, Hillsborough County.

Drainage area.--650 sq mi, approximately.

Records available.--October 1938 to September 1960.

Gage.--Water-stage recorder. Datum of gage is at mean sea level, datum of 1929 (city of Tampa bench mark). Prior to Oct. 1, 1945, at site 1.4 miles upstream at datum 0.66 ft higher.

Average discharge.--22 years (1938-60), 685 cfs (495,900 acre-ft per year), adjusted for diversion.

Extremes.--1938-60: Maximum discharge, 14,600 cfs Mar. 21, 1960; maximum gage height, 22.89 ft Aug. 2, 1960; no flow Nov. 30 to Dec. 2, 1945.
Maximum stage known, 25.6 ft Sept. 7, 1933, at former site and datum, from floodmarks, affected by backwater prior to failure of Tampa power dam, 1.4 miles below former gage. A discharge of 16,500 cfs was measured Sept. 9, 1933.

Remarks.--Flow regulated by Tampa reservoir since Oct. 1, 1945. Capacity of reservoir insufficient to affect monthly figures of runoff. Diversion at point 1½ miles above station for water supply by city of Tampa. Records of chemical analyses for the period November 1956 to September 1958 are published in reports of the Geological Survey.

Monthly and yearly mean discharge, in cubic feet per second a/

Water year	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	The year
1951	615	236	364	353	234	199	406	186	52.4	144	498	697	333
1952	751	221	358	185	169	521	598	80.1	257	226	644	429	371
1953	787	391	141	188	321	171	1,065	217	170	469	1,965	4,371	852
1954	2,795	640	1,795	850	250	177	103	97.8	327	723	1,199	308	781
1955	268	109	129	125	216	76.6	65.1	33.1	34.3	215	800	1,099	264
1956	191	197	120	83.1	174	42.9	13.9	14.8	8.48	31.5	35.9	324	102
1957	742	172	46.1	34.8	49.4	366	759	388	318	528	1,834	1,790	588
1958	1,348	157	119	414	486	1,975	847	204	74.5	494	811	238	601
1959	144	170	134	887	417	3,082	2,022	740	1,853	2,704	2,738	3,597	1,546
1960	1,957	805	217	231	464	4,926	1,358	154	220	1,200	4,713	4,276	1,718

a Unadjusted for diversion by city of Tampa.

Monthly and yearly discharge, in acre-feet

Water year	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	The year
1951	37,830	14,040	22,400	21,710	13,010	12,240	24,160	11,410	3,120	8,870	30,600	41,450	240,800
1952	46,180	13,160	22,020	11,390	9,700	32,030	35,560	4,920	15,270	13,910	39,670	25,530	269,300
1953	48,410	23,270	8,680	11,590	17,820	10,490	63,360	13,320	10,140	28,830	120,900	260,100	616,900
1954	171,900	38,070	110,400	52,280	13,880	10,880	6,150	6,010	19,440	44,450	73,720	18,330	565,500
1955	16,500	6,510	7,940	7,690	11,980	4,710	3,870	2,040	2,040	13,190	49,190	65,400	191,000
1956	11,730	11,730	7,370	5,110	10,030	2,640	830	908	505	1,940	2,210	19,280	74,280
1957	45,640	10,250	2,830	2,140	2,750	22,490	45,140	23,880	18,920	32,470	112,800	106,500	425,800
1958	82,870	9,350	7,320	25,440	27,000	121,500	50,380	12,520	4,440	30,380	49,890	14,170	435,300
1959	8,850	10,110	8,230	54,530	23,180	189,500	120,300	45,520	110,200	166,300	168,300	214,000	1,119,000
1960	120,400	47,900	13,370	14,230	26,700	302,900	80,840	9,440	13,070	73,770	289,800	254,400	1,247,000

Note.--Figures given herein prior to October 1956, not previously published.

Yearly discharge, in cubic feet per second

<table>
<tr><th rowspan="4">Year</th><th rowspan="4">WCP</th><th colspan="8">Water year ending Sept. 30</th><th colspan="4">Calendar year</th></tr>
<tr><th colspan="5">Observed</th><th colspan="3">Adjusted a/</th><th colspan="2">Observed</th><th colspan="2">Adjusted a/</th></tr>
<tr><th colspan="2">Maximum day</th><th rowspan="2">Minimum day</th><th rowspan="2">Mean</th><th rowspan="2">Acre-feet</th><th rowspan="2">Mean</th><th rowspan="2">Per square mile</th><th rowspan="2">Runoff in inches</th><th rowspan="2">Mean</th><th rowspan="2">Acre-ft</th><th rowspan="2">Mean</th><th rowspan="2">Runoff in inches</th></tr>
<tr><th>Discharge</th><th>Date</th></tr>
<tr><td>1950</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>-</td><td>498</td><td>#360,700</td><td>526</td><td>10.99</td></tr>
<tr><td>1951</td><td>1204</td><td>1,500</td><td>Sept.24,25,1951</td><td>24</td><td>333</td><td>#240,800</td><td>361</td><td>0.555</td><td>7.53</td><td>342</td><td>#247,900</td><td>370</td><td>7.74</td></tr>
<tr><td>1952</td><td>1234</td><td>1,900</td><td>Apr.5,1952</td><td>28</td><td>371</td><td>#269,300</td><td>399</td><td>.614</td><td>8.36</td><td>370</td><td>#268,300</td><td>[illegible]</td><td>8.33</td></tr>
<tr><td>1953</td><td>1274</td><td>6,830</td><td>Sept.30,1953</td><td>37</td><td>852</td><td>#616,900</td><td>882</td><td>1.36</td><td>18.42</td><td>1,183</td><td>#856,900</td><td>1,213</td><td>25.33</td></tr>
<tr><td>1954</td><td>1334</td><td>b2,890</td><td>July 31,1954</td><td>28</td><td>781</td><td>#565,500</td><td>810</td><td>1.25</td><td>16.91</td><td>381</td><td>#276,000</td><td>[illegible]</td><td>8.57</td></tr>
<tr><td>1955</td><td>1384</td><td>1,980</td><td>Sept.15,1955</td><td>30</td><td>264</td><td>#191,000</td><td>296</td><td>.455</td><td>6.16</td><td>264</td><td>#190,900</td><td>296</td><td>6.18</td></tr>
<tr><td>1956</td><td>1434</td><td>810</td><td>Sept.28,1956</td><td>6.4</td><td>102</td><td>#74,280</td><td>138</td><td>.212</td><td>2.91</td><td>141</td><td>102,200</td><td>[illegible]</td><td>3.73</td></tr>
<tr><td>1957</td><td>1504</td><td>3,810</td><td>Aug. 10,1957</td><td>20</td><td>588</td><td>425,800</td><td>623</td><td>.958</td><td>13.01</td><td>645</td><td>466,600</td><td>[illegible]</td><td>14.20</td></tr>
<tr><td>1958</td><td>1554</td><td>3,180</td><td>Oct. 6,1957</td><td>31</td><td>601</td><td>435,300</td><td>638</td><td>.982</td><td>13.34</td><td>501</td><td>362,900</td><td>[illegible]</td><td>[illegible]</td></tr>
<tr><td>1959</td><td>1624</td><td>7,390</td><td>Mar.23,24,1959</td><td>84</td><td>1,546</td><td>1,119,000</td><td>1,586</td><td>2.44</td><td>33.11</td><td>1,759</td><td>1,274,000</td><td>1,799</td><td>37.57</td></tr>
<tr><td>1960</td><td>1704</td><td>c14,600</td><td>Mar. 21,1960</td><td>17</td><td>1,718</td><td>1,247,000</td><td>1,760</td><td>2.71</td><td>36.84</td><td>-</td><td>-</td><td></td><td>-</td></tr>
</table>

# Not previously published.

a Adjusted for diversion by city of Tampa; diversion records furnished by city of Tampa Water Department.

b Maximum daily discharge for flood event whose crest occurred in the water year indicated. Maximum daily discharge, 6,800 cfs Oct. 1, 1953, on the recession from the crest that occurred in the preceding water year.

c Momentary maximum.

STATISTICS OF RECEIVING WATER RESPONSE TO RUNOFF

By

Dominic M. DiToro*
Environmental Engineering and Science Program
Manhattan College, Bronx, New York

Abstract

An analysis is presented for the mean and variance of a one dimensional advective dispersive system that is subjected to random inputs of runoff. Analytical solutions are available for which the mass input is represented as a Poisson process of delta functions. The effects of event to event variability of runoff mass discharges are included in the formulation as are the random times between overflows. The method of solution evaluates the overlapping effects in the receiving water due to the persistence of discharged mass. Both conservative, first order reactants, and sequentially reacting substances are considered.

The solutions have certain unexpected properties. In particular, the normalized variance of BOD and DO are symmetric about the discharge point, whereas the means of the concentration are not. This is explained in terms of the effects of advection and dispersion of fluctuations. The analytical solutions are compared to simulated results using an observed hourly rainfall sequence. The results indicate that the within event variability is not significant, if the receiving water dispersion is large enough. It is also pointed out that treatment devices such as retention basins which remove a certain average fraction of the overflowing mass are less effective in removing variance in the receiving water. This phenomena is of importance in the evaluation of the probable benefit of runoff treatment.

1. Introduction

The analysis of receiving water responses to intermittent discharges from separate and combined sewer overflows is a problem of substantial difficulty. The problem setting is inherently time variable due to the transient mass discharges, and, although a time variable integration of the mass balance equations describing the receiving water response is possible, it is an unwieldy and laborious undertaking.

---

*Consultant, Hydroscience, Inc., Westwood, N.J., Associate Research Professor Environmental Engineering and Science Program. Manhattan College, Bronx, N.Y. 10471.

Consider the results of such a simulation: receiving water concentrations are calculated for each volume element for the duration of the integration, e.g. three summer months. And since "the purpose of computing is insight not numbers" [1] it is necessary to perform this calculation many times with different mass discharge sequences, treatment alternatives, and receiving water parameters in an effort to determine the important controlling mechanisms. Faced with this large amount of information, it is probable that the computational results would be summarized in terms of the relevant statistics: the mean, variance, and the percentiles of the receiving water concentration.

The purpose of this paper is to present a statistical analysis of the response of a water body to a sequence of runoff mass discharges. The statistics of the receiving water concentration are related to the statistics of the runoff process and the parameters of the water body. Certain quite general formulas are presented which apply to any receiving water type. These are specialized for an advective-dispersive system and analytical solutions are derived for the mean and variance of the response.

2. Runoff as a Pulse Process

Consider a particular discharge event i with duration d_i, centered at time t_i, for which the mass discharge rate within the event is constant, w_i. Then the mass discharge rate entering the receiving water for the period of the analysis, W(t), is the superposition of these events, suitably displaced in time:

$$W(t) = \sum_i w_i \, r(d_i, t-t_i) \qquad (1)$$

where:

$$r(d_i,t) = \begin{cases} 1 & -d_i/2 \le t \le d_i/2 \\ 0 & \text{elsewhere} \end{cases} \qquad (2)$$

a rectangular pulse with unit height and duration d_i, centered at time zero.

If the substance being considered is conservative or decays following linear kinetics, e.g. total suspended solids, coliforms, or BOD, then the principle of superposition applies and the receiving water response, c(x,t), at any location, x, can be written as the sum of the responses to the individual mass discharges suitably displaced in time:

$$c(x,t) = \sum_i w_i \, h(x,t-t_i,d_i,q_i) \qquad (3)$$

where h is the unit response of the receiving water to a discharge centered at time t_i, of unit magnitude (w_i=1), duration d_i, with discharge flow, q_i, augmenting the flow in the receiving water. The situation is illustrated in Figure 1. The sequence of mass discharges produces a sequence of individual concentration responses which possibly overlap and their superposition is the receiving water response. The problem is to compute the statistics of c(x,t) from the statistics of the runoff and the parameters that determine the unit response of the receiving water.

The analysis is tractable if the occurrence times, t_i, satisfy certain assumptions. These are best described in terms of δ_i, the time between the midpoints of the events:

$$\delta_i = t_{i+1} - t_i \qquad (4)$$

It will be assumed in what follows that:

(a) δ_i are exponentially distributed, that is:

$$\Pr\{\delta_i > \delta\} = \exp(-\delta/\Delta) \qquad (5)$$

where Δ is the average time between the midpoints of the events. This distribution has the property that the standard deviation of δ equals the mean so that the coefficient of variation: $\nu_\delta = \sigma_\delta/\Delta = 1$;

(b) the time between events are serially uncorrelated and in fact independent, that is:

$$\rho_\delta(k) = E\{\frac{\delta_i-\Delta}{\sigma_\delta} \frac{\delta_{i+k}-\Delta}{\sigma_\delta}\} = 0 \qquad k > 0 \qquad (6)$$

These assumptions imply that t_i is a Poisson process. In addition the other runoff parameters must satisfy certain assumptions:

(c) The times between events are uncorrelated with the other runoff variables, that is

$$\rho_{d\delta} = E\{\frac{d_i-D}{\sigma_d} \frac{\delta_i-\Delta}{\sigma_\delta}\} = 0 \qquad (7)$$

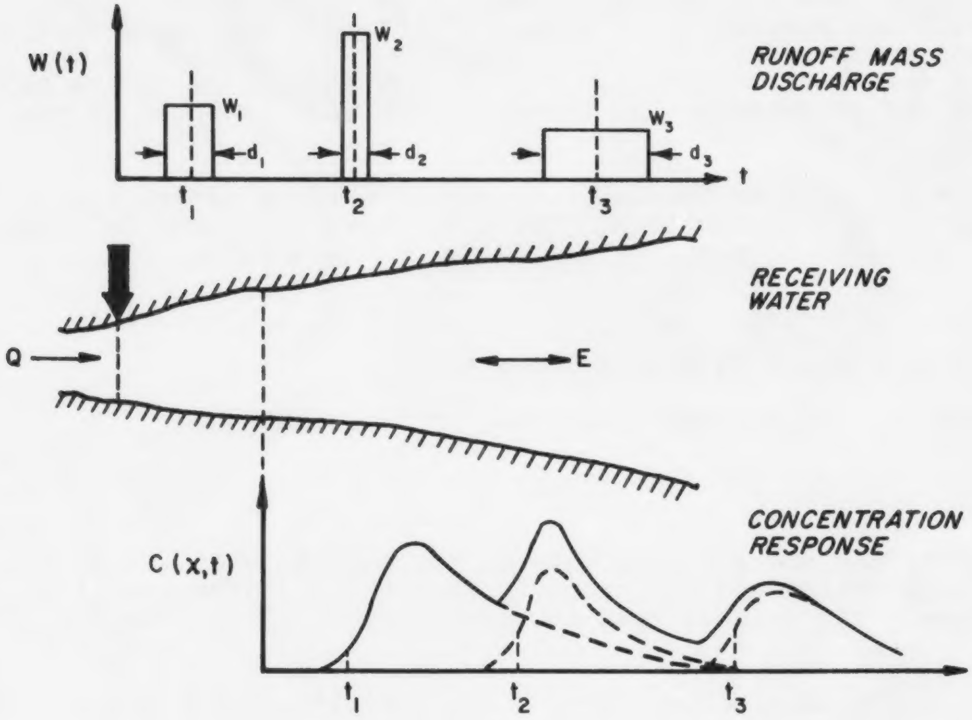


Figure 1. Schematic representation of the problem framework. Mass discharges corresponding to runoff events at t_1, t_2,..... enters the receiving water at $X = 0$. The resulting concentration response is the superposition of the individual concentration responses.

where $D = E\{d_i\}$; and similarly for $\rho_{q\delta}$ and $\rho_{w\delta}$.

(d) The runoff quantities: w_i, d_i and q_i are serially uncorrelated:

$$\rho_d\,(k) = E\{\frac{d_i - D}{\sigma_d}\,\frac{d_{i+k} - D}{\sigma_d}\} = 0 \qquad k > 0 \qquad (8)$$

and similarly for w_i and q_i.

(e) The probability distributions of all the random variables are independent of time (stationarity).

(f) The principle of superposition applies and the receiving water response is a function only of the time difference $t-t_i$, and not an explicit function of the time t_i. That is, the coefficients in the relevant mass balance equations are constants in time (time invariant linear system).

Although it is difficult to verify assumptions (a)-(e) with observed runoff data since they are usually of insufficient quantity, it is straightforward for rainfall data. The comparisons can be made using standard goodness of fit tests. As an example of the procedure, the hourly rainfall data for the summer of 1967 (an average summer) for Boston's Logan Airport gauge are converted into event variables [2]. The choice of the minimum length of dry hours that separate storm events is based on assumption (a). In particular it is chosen so that $\nu_\delta \tilde{=} 1$. This is achieved for this example if five consecutive dry hours is used. The statistics of the event variables are shown in Table 1. A more stringent test is to compare the empirical and theoretical log survivor function, which is the log of one minus the cumulative distribution function of the midpoint interval, Eq. 5 [3]. As shown in Fig. 2 the empirical survivor function is nearly exponential with a slight over-representation of small midpoint intervals. The straight line in Fig. 2 is the theoretical curve using $\Delta = 2.70$ days from Table 1, displaced downward by 0.5 units for visual clarity.

Assumptions (b) and (d), the serial independence of event variables, are tested using the sample autocorrelation coefficients which are Eqs. (6) and (8) with averages replacing the expectations. These are shown in Fig. (3) with their approximate 80% confidence limits [4]. Although the lag one and two autocorrelations are slightly negative, indicating

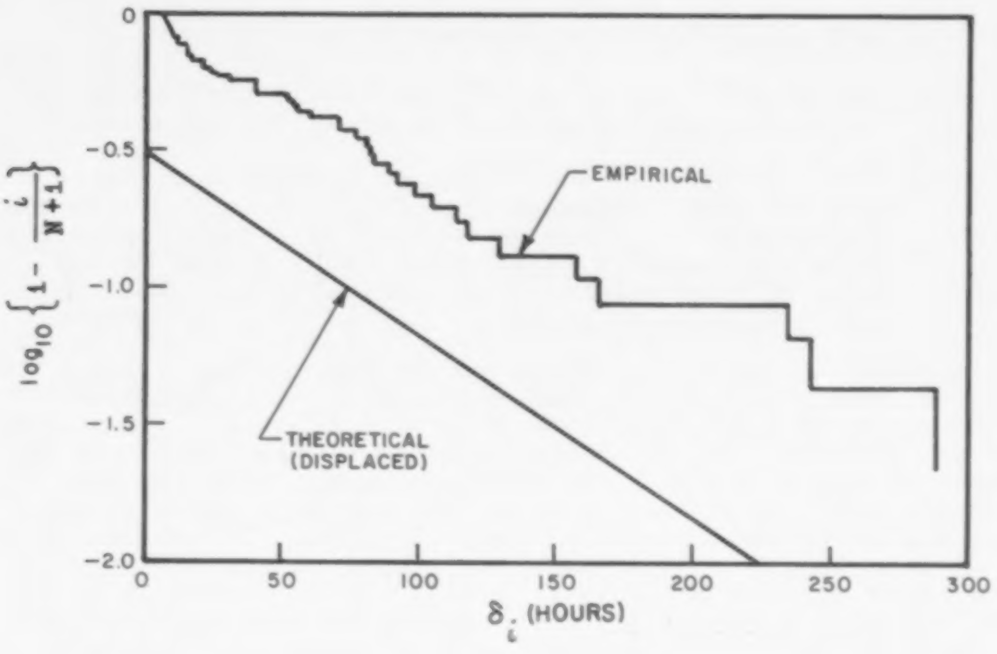


Figure 2. Empirical log survivor function of the midpoint intervals. The ordinate is $\log_{10}$ of the plotting position; the abcissa are the values of the ordered observed midpoint intervals. The theoretical curve (displaced downward by 0.5 for visual clarity) represents the result for an exponentially distributed random variable.

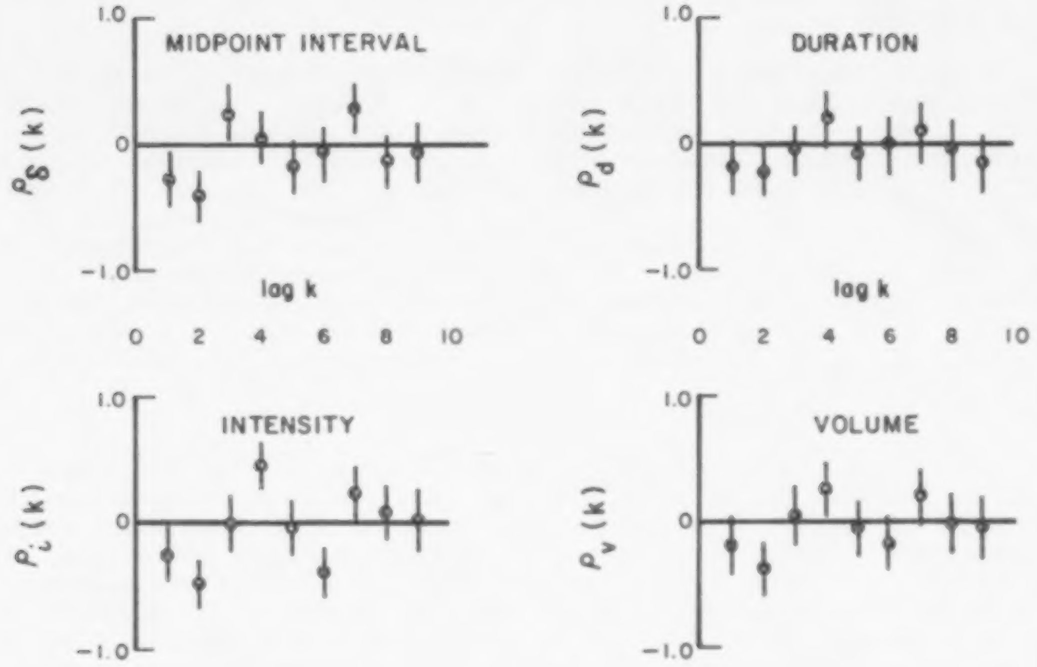


Figure 3: Sample serial auto correlations for storm midpoint interval, duration, intensity, and volume with approximate 80% confidence limits.

there is a slight tendency for large values to be followed by smaller values of the event variables, this effect is small and the independence assumption appears reasonable.

Assumption (c), the lack of cross correlation between the midpoint interval and the other event variables is tested in the same way. Table 1 presents the results: $\rho_{i\delta}$ and $\rho_{v\delta}$ are not significantly different from zero; however $\rho_{d\delta} = 0.448$ is significant (99%). This would affect the analysis if the receiving water response is strongly affected by the runoff duration. As shown subsequently the more important quantity is runoff volume which is not significantly correlated to the midpoint interval.

Perhaps the most difficult assumption to verify is that $\rho_{w\delta} = 0$ since this requires actual runoff data. As shown in Table 1, the rainfall volume is independent of the preceding midpoint interval. However, lack of correlation between w_i and δ_i requires that the concentration, c_i, of the runoff also be uncorrelated with δ_i. This is a site specific requirement that depends on the mechanisms which influence the resultant pollutant concentrations. If the principle mechanism is a steady buildup of pollutants on the land surface between runoff events then a strong correlation might be expected. However, in one case at least [5] only a slight relationship was observed. In any case this assumption provides a convenient starting point for the analysis of receiving water response.

The stationarity assumptions (e) and (f) can be violated if the period of analysis spans too long a period of time since rainfall has definite seasonal patterns, as does receiving water temperature and flow. However, stationary periods for runoff can be chosen from an inspection of the monthly distributions of the relevant rainfall statistics. An example [2] is shown in Fig. 4. In particular it can be seen that the change in the means and coefficients of variation over a three month summer period are not too severe so that stationarity is a reasonable assumption for certain periods of time. This period should also coincide with the period for which the receiving water parameters are constant, assumption (f).

In summary, these assumptions guarantee that the runoff events occur purely randomly in time; that their properties are uncorrelated event to event; and that the seasonal effects are negligible. It is for this simplified yet realistic situation that analytical results are available.

TABLE 1

RAINFALL STORM STATISTICS
LOGAN AIRPORT, BOSTON, MASSACHUSETTS

May 15 - September 15, 1967

	δ (day)	i (in/hr)	d (hr)	v (in)
Mean	2.70	0.0706	4.73	0.358
ν	1.04	1.32	1.22	1.98

Cross Correlation Coefficients

δ	1.0	-0.149	0.448	0.257
i		1.0	0.044	0.592
d			1.0	0.735

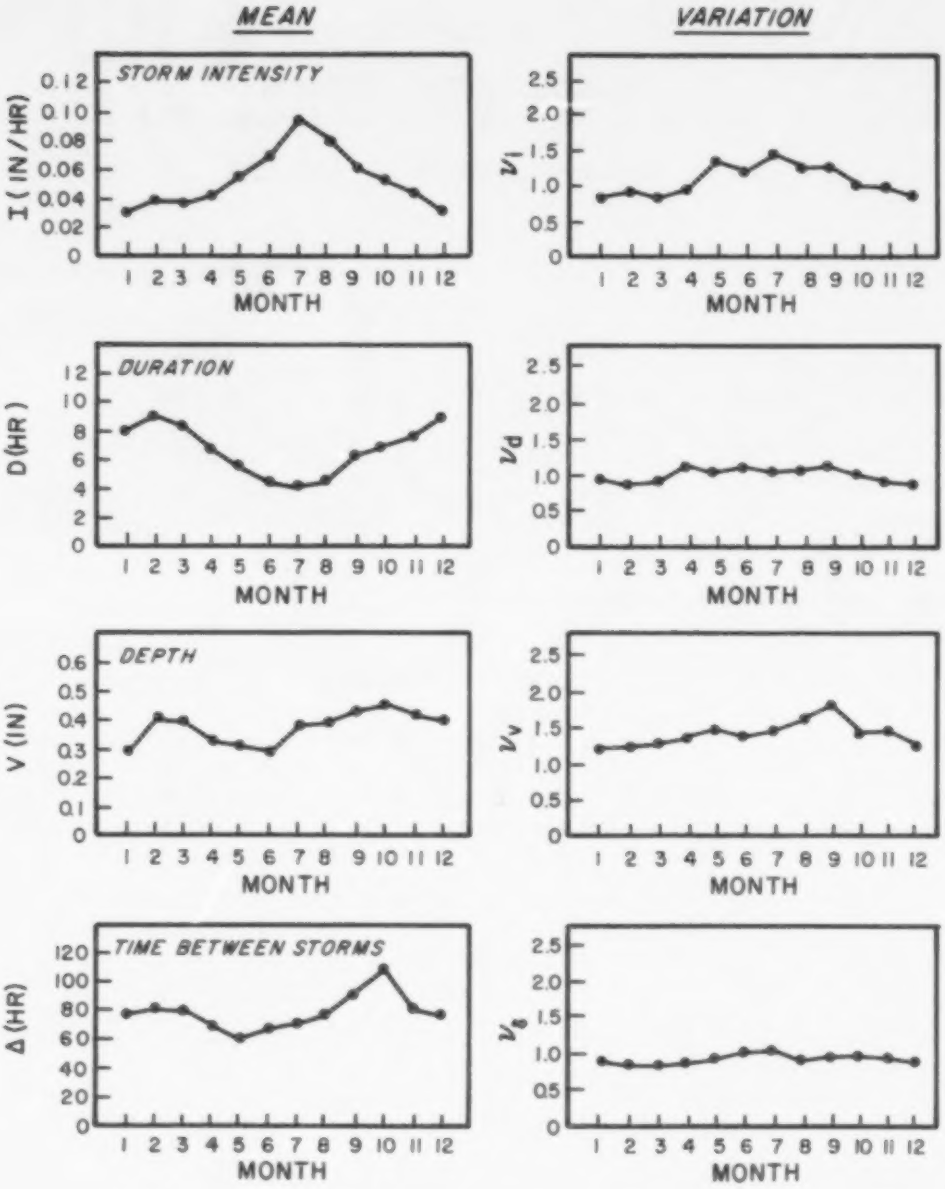


Figure 4. Seasonal variation of the means and coefficients of variation for storm intensity, duration, volume, and midpoint integral from the Central Park, New York City rain gauge for the years 1948-1975. Values are computed from all storms occurring within each month.

3. Mean and Variance of the Concentration Response

The expressions for the mean and variance of the concentration, Eq. (2), are available from an application of the Generalized Campbell Theorem for Poisson processes [6],[7]. Applying Campbell's theorem to Eq. (3) yields these general results for the receiving water concentraton mean and variance:

$$E\{c(x,t)\} = \frac{1}{\Delta} E\{w_i \int_{-\infty}^{\infty} h(x,t,d_i,q_i)\, dt\} \qquad (9)$$

$$V\{c(x,t)\} = \frac{1}{\Delta} E\{w_i^2 \int_{-\infty}^{\infty} h^2(x,t,d_i,q_i)\, dt\} \qquad (10)$$

These remarkable formulas are a complete solution for the statistics of the receiving water concentration, the problem posed in the introduction. It is only necessary to compute the time integrals of the unit response and its square, and then to perform the expection (i.e. the statistical average) of these quantities with respect to the runoff variables: w_i, d_i and q_i. This is an important simplification of the alternative: to compute long term simulations for the period of interest and statistically analyze the result. Instead, what is required is the unit responses for a sufficient range of durations and runoff flows so that the expectations can be computed. For the idealized case in which the duration and runoff flow do not significantly affect the unit response, then only a single unit response is necessary. This case is discussed below.

These equations provide a method for investigating the importance of various phenomena on the mean and variance. The question can now be posed in terms of what effect these mechanisms have on the time integrals and expectations in the formulas. In particular, it is immediately apparent that increasing the average time between events decreases both the mean and the variance as $1/\Delta$. This is the case regardless of the relative decrease in the overlapping effects of successive runoff events as the time between events increase.

Such insights are valuable even if complete simulations are computed since if the simulations behave in a significantly different way, then the analyst can inquire into the cause of this unexpected behavior. In fact these formulas provide a basis for the intuitive understanding of receiving water response to random pulse loadings. Such information is invaluable when dealing with complex numerical simulations since the question: are the simulations correct, is of constant concern, and methods of confirming the computations are constantly sought. Eqs. (9) and (10) provide the basis for a validation of the simulation.

4. Application to Advective-Dispersive System

The equations for the mean and variance, Eqs. (9) and (10) can be evaluated directly if the unit response of the receiving water is independent of the runoff duration and flow. The latter is the case in rivers and estuaries for which the base freshwater flow is large relative to the runoff flow. The effect of duration is small if the dispersion in the receiving water is large. For moderate dispersion the effect of finite duration runoff events is to spread out the loading over time and reduce the peak receiving water concentration. Using the approximation discussed below will tend to overestimate the receiving water impact in this case and provides an upper bound.

Consider the limiting case for which the duration approaches zero and the mass discharge rate approaches infinity in such a way that the mass discharged at each event, m_i, is finite. The result is a mass discharge rate of the form:

$$W(t) = \sum_i m_i \, \delta(t-t_i) \tag{11}$$

where $\delta(t)$ is a delta function: an impulse of infinite height, zero width, and unit area centered at $t = 0$. The receiving water response is:

$$c(x,t) = \sum_i m_i \, h(x,t-t_i) \tag{12}$$

where $h(x,t)$ is the unit impulse response of the receiving water. These equations are limiting versions of Eqs. (1) and (3) for duration approaching zero and no runoff flow effect. The version of Campbell's theorem that applies in this case is:

$$E\{c(x,t)\} = \frac{M}{\Delta} \int_0^\infty h(x,t)\, dt \tag{13}$$

$$V\{c(x,t)\} = \frac{M^2(1+\nu_M^2)}{\Delta} \int_0^\infty h^2(x,t)\, dt \tag{14}$$

where $M = E\{m_i\}$, the mean mass discharged, and $\nu_M^2 = V\{m_i\}/E^2\{m_i\}$, the normalized variance of the mass discharged at each event.

Eq. (13) for the mean has a direct and general interpretation. The time integral of the impulse response is the steady state response to a continuous unit mass discharge rate. $E\{m_i\}/\Delta$ can be thought of as the mean mass discharge rate (mean mass discharged during an event/mean time between events). Hence, Eq. (13) implies that the average receiving water concentration can be calculated from the steady state response to a continuous constant mass discharge corresponding to the mean runoff mass discharge rate. This general result applies to any receiving water configuration - the form of $h(x,t)$ is as yet unspecified - so that the results of steady state numerical model calculations are directly applicable in computing the average concentration response, if runoff duration and flow effects are negligible.

Unfortunately the integral of the square of the impulse response has no general interpretation and it must be computed or approximated in each case. The expression involving the mass discharge statistics is interesting since it indicates that the variance of the receiving water concentration is due to two effects: the intermittent discharges, and the event-to-event variability of the mass discharged. Even if the mass discharged at each event were of equal magnitudes so that $\nu_M = 0$, there would still be receiving water concentration variance due to the intermittent nature of the discharges. Event-to-event variation of mass discharges augments the variance as shown in Eq. (14).

The magnitude of this effect can be estimated in the following way. Since $m_i = c_i v_i$, and if it is assumed that c_i and v_i are independent, then:

$$\nu_M^2 = \nu_c^2 \nu_v^2 + \nu_v^2 + \nu_c^2 \qquad (15)$$

For the limiting case of constant concentration, $\nu_c = 0$ and $\nu_M^2 = \nu_v^2$ which from Table 1 is seen to be $\nu_v = 1.98$ for this example so that $\nu_M^2 \simeq 4$ and the effect of event-to-event variation is to increase the variance by a factor of five, which is quite substantial.

5. Constant Parameter Case

In order to compute the square of the impulse response a specific receiving water configuration is required. For the case of an advective-dispersive system with constant parameters, the impulse response to a unit mass discharge is well known to be [8]:

$$h(x,t) = \frac{1}{A\sqrt{4\pi E t}} \exp\left\{-\frac{(x-Ut)^2}{4Et} - Kt\right\} \quad t \geq 0 \qquad (16)$$

where A, E, U, K are the cross-sectional area, dispersion, velocity, and decay coefficients respectively. The evaluation of the mean and variance involves the definite integral [9]:

$$\int_0^\infty e^{-\lambda t}\, t^{\nu-1}\, e^{-\alpha/4t}\, dt = 2\left(\frac{\alpha}{4\lambda}\right)^{\nu/2} K_\nu(\sqrt{\alpha\lambda}) \qquad (17)$$

where $K_\nu(Z)$ is the modified Bessel function [10] with the special case $K_{\frac{1}{2}}(Z) = \sqrt{\pi/2Z}\, e^{-Z}$. For the case of the mean the result is:

$$E\{c(x,t)\} = \frac{M}{\Delta UAm}\, e^{\frac{p}{2}(1 \mp m)} \qquad (18)$$

where $p = Ux/E$, $\eta = KE/U^2$, and $m \;\sqrt{1+4\eta}$. The minus sign applies for $x > 0$ and the positive sign for $x < 0$. As expected, this is the steady state solution for an equivalent continuous mass discharge rate of M/Δ.

The variance of the concentration follows directly from Eqs. (14) and (17):

$$V\{c(x,t)\} = \frac{M^2(1+\nu_M^2)}{2\pi EA^2\Delta}\, e^p\, K_o(mp) \qquad (19)$$

where since $K_o(-Z) = K_o(Z)$ Eq. (19) applies for all x.

The behavior of the variance at the point of discharge, $x = 0$, is unrealistic since it is infinite, $K_o(Z) \sim -\ln(Z)$ as $Z \to 0$. The mathematical reason is that the impulse response at $x=0$ is $h(0,t) \sim e^{-Kt}/\sqrt{t}$ and the integral of the square of this function is infinite. This is due to the impulse of zero duration and infinite mass discharge rate which produces an infinite concentration at the moment of discharge. The concentration then decays slowly enough to cause the variance of concentration to become infinite. In fact the smoothing induced by the finite duration of the runoff event is not included in the analysis and the unrealistic result is not too surprising.

An unexpected consequence of these equations appears if the normalized variance of concentration is considered:

$$\nu^2\{c(x,t)\} = \Delta(1+\nu_M^2)\, \frac{U^2m^2}{2\pi E}\, e^{m|p|}\, K_o(mp) \qquad (20)$$

where $\nu^2\{c(x,t)\} = V\{c(x,t)\}/E^2\{c(x,t)\}$. The normalized variance is symmetric about the point of discharge whereas neither the mean nor the variance are symmetric if the freshwater flow is significant. Fig. 5 illustrates the spatially varying portion of the solution for a hypothetical BOD discharge into an estuary. The asymmetry of the mean response is in sharp contrast to the symmetry of the normalized variance. The infinite variance at the origin is also apparent although it is clear that it decreases rapidly as the distance from the origin increases.

The reason for this symmetry is related to the effect of the advective portion of the mass transport on the mean and variance. The effect of pure advection and decay on a time variable mass discharge, $W(t)$, is to translate the time axis by the travel time x/U, i.e. $c(x,t) = W(t-x/U) \exp(-Kx/U)/UA$ with an appropriate reduction due to decay [11]. Thus the normalized variance in the receiving water at any point downstream of the discharge, assuming no runoff flow effect, is the normalized variance of the mass discharge. Advection, by itself, provides no reduction in normalized variance. It is dispersion that provides the smoothing that reduces normalized variance. And since the effect of dispersion is increased as the distance either upstream or downstream from the discharge point increases, the normalized variance decreases symmetrically from the point of discharge. In effect the advection compresses the upstream profile and expands the downstream profile but the contraction and expansion affects both the mean and variance. The normalized variance removes this effect and exhibits only the symmetrical effects of the smoothing caused by the symmetrical dispersion.

The effect of increasing the average time between events, Δ, is to increase the normalized variance. The reason is that the concentration as a function of time is more variable relative to the mean since, although the events occur less frequently, they still produce the same spikes of concentration in the receiving water. Conversely as the time between events is decreased the events occur more and more frequently with more and more overlap until the discharge approaches a continuous, constant, discharge rate and the normalized variance approaches zero.

The behavior of the normalized variance as a function of position and reaction rate is illustrated in Fig. 6: a contour plot of $\nu^2\{c(x,t)\}/[(1+\nu_M^2)U^2\Delta/E]$. The dimensionless parameters for position, Ux/E, reaction rate, KE/U^2, and the normalization have been chosen to isolate the effect of varying the reaction

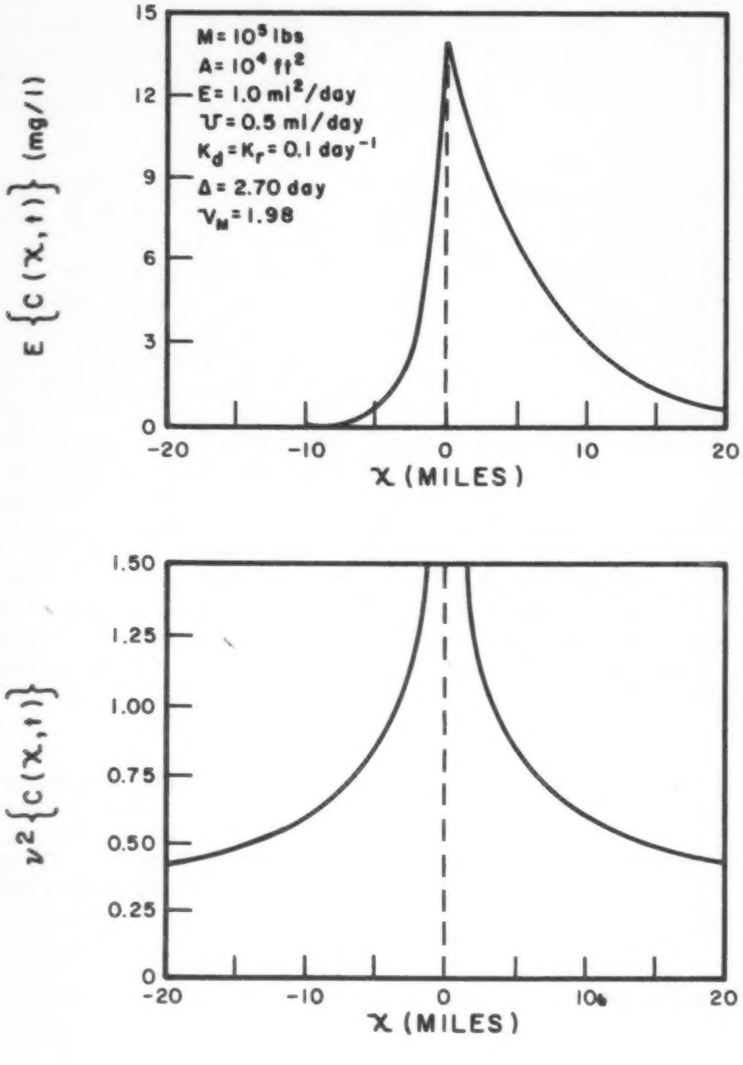


Figure 5. Computed mean and normalized variance for BOD discharged into an hypothetical estuary.

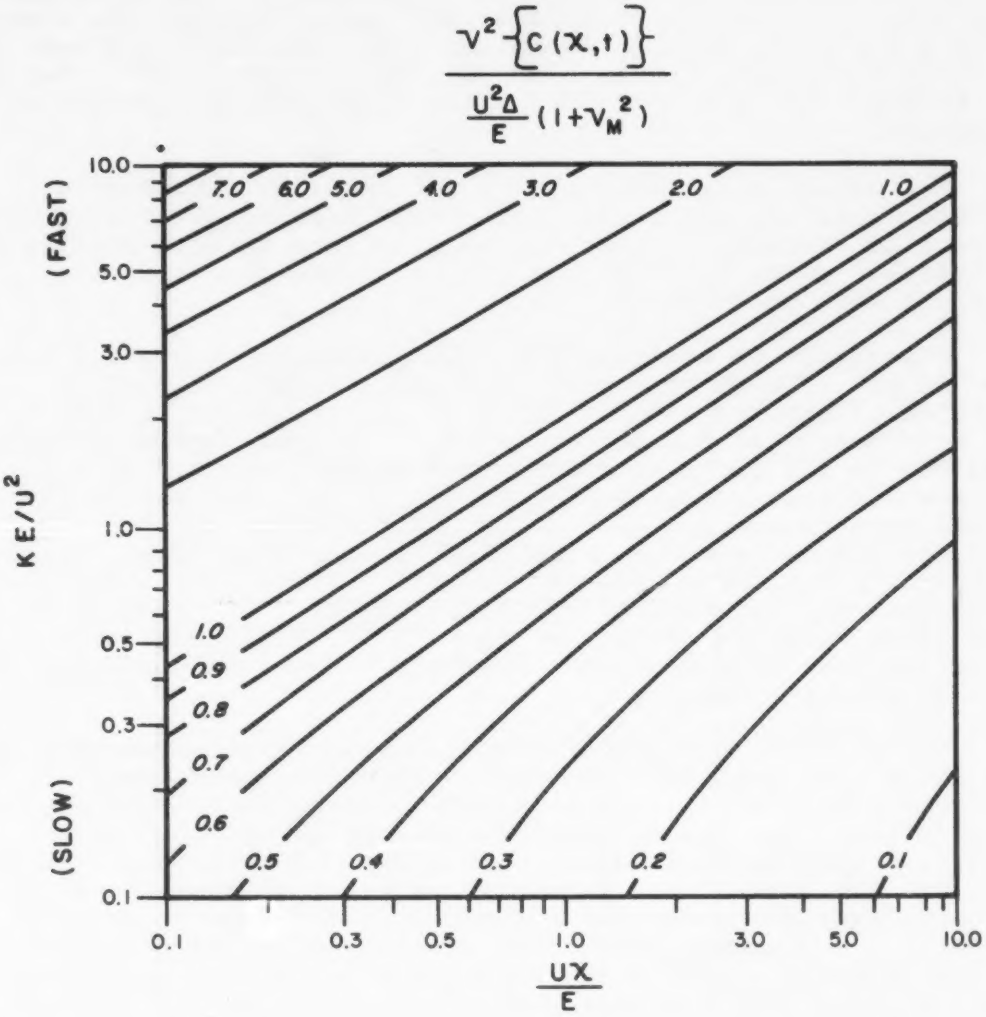


Figure 6: Contour plot of dimensionless normalized variance versus normalized distance and estuary number.

rate. Increasing the reaction rate increases the normalized variance: e.g. coliform discharges produce much higher normalized variance then do slowly reacting substances, e.g. BOD. The reason is that the initial concentration in the receiving water due to a runoff event is independent of decay rate whereas the average concentration is markedly influenced. As a result the variance caused by the high initial concentrations remains as the reaction rate increases whereas the mean concentration decreases.

The almost parallel contours indicate that increasing reaction rate increases the normalized variance uniformly with respect to distance so that the shape of the spatial profile is essentially unchanged and only the magnitude increases as reaction rate increases.

The effect of changing the transport characteristics can be seen in Fig. 7 a contour plot of $\nu^2\{c(x,t)\}/K\Delta(1+\nu_M^2)$ as a function of estuary number, KE/U^2, and position as indexed by the mean concentration profile. The locations considered range from 90% to 10% of the concentration at the origin. This distance scale tracks the spatial extent of the relevant portion of the mean concentration profile. A number of interesting features are illustrated. For $KE/U^2>1$, the normalized variance is essentially independent of KE/U^2, the ratio of dispersion to advection. For a nine fold reduction in mean concentration, normalized variance is reduced three fold, indicating that normalized variance decreases less slowly than mean concentration in this spatial region.

For $KE/U^2<1$, dispersion is decreasing relative to advection and normalized variance increases. This four fold spatial decline at $KE/U^2=0.1$ is more marked than the three fold decline at $KE/U^2=10.0$ but the differences are not large, indicating that, for this spatial region, the effect of changes in transport are not excessively dramatic: a two fold or less change in normalized variance over a hundred fold change in estuary number. The somewhat surprising conclusion is that reaction kinetic effects are much more important in determining normalized variance than are transpart effect so long as $KE/U^2>0.1$, and runoff flow and finite duration effects are small.

6. Comparison to Simulations

A critical assumption in the above analysis is that the mass for each runoff event enters the receiving water as a narrow pulse. In fact, the mass is discharged over a time interval that is comparable to the duration of each rainfall

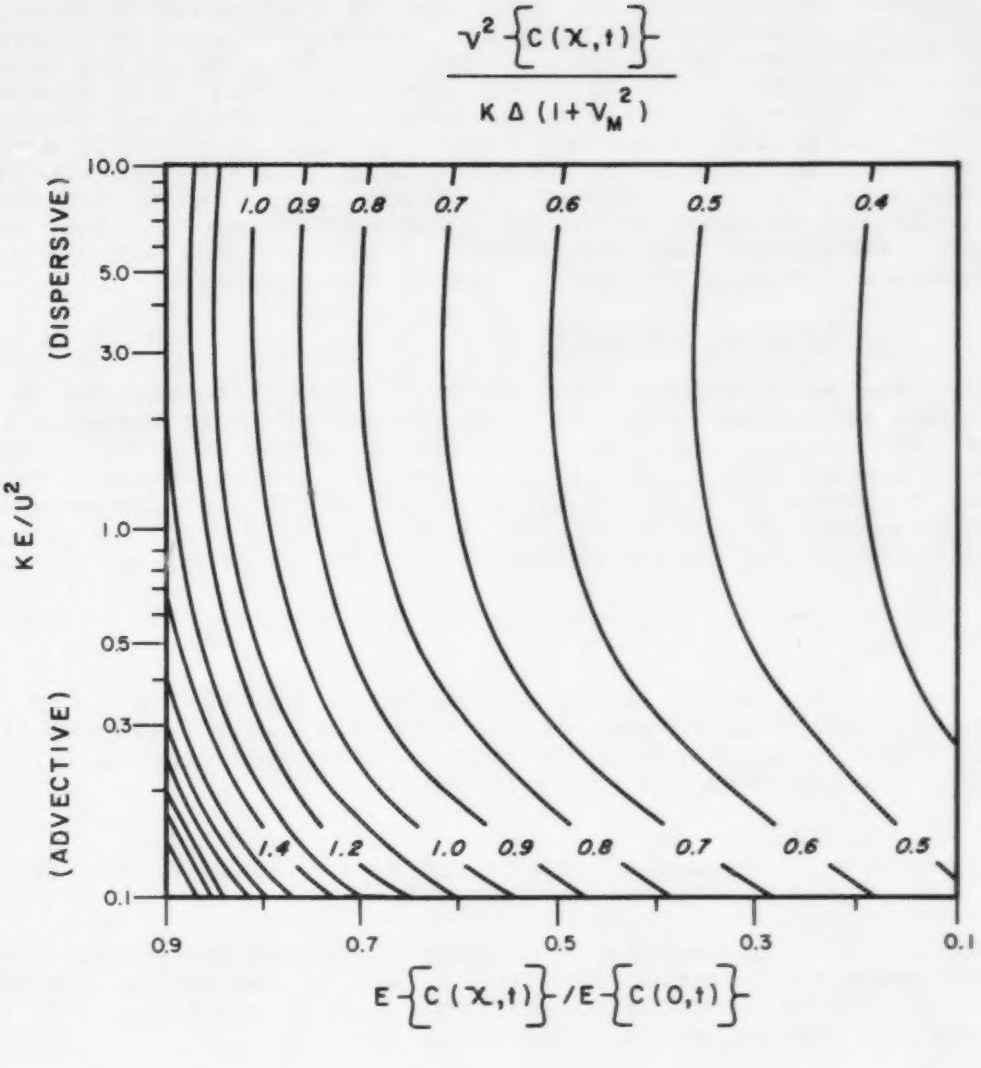


Figure 7: Contour pilot of dimensionless various versus fraction of concentrations at the origin and estuary number.

event. In order to check the validity of this simplification, a time variable simulation of a two month summer period has been carried out [2]. The parameters of the advective dispersive system are chosen to be representative of the Hudson River estuary during summer low flow. A single point source is used in order to highlight any difficulties that might be masked if multiple inputs were considered. The runoff mass is discharged in accordance with the hourly rainfall observed at the Central Park raingage. A constant coliform concentration is assumed. However the mass discharge has both within-event and event-to-event variability since the runoff volumes vary hourly within the event and from event to event in accordance with the raingage data. A comparison of the theoretical results and the simulation is shown in Fig. 8. The results are in close agreement, indicating that the effect of within-event variation and overflow duration are negligible in the estuary.

7. Dissolved Oxygen Deficit

The methods developed in the previous sections can be extended to sequentially reacting substances, most noteably BOD and DO deficit. All that is required is the DO deficit impulse response to an impulse of BOD. This is easily obtained from the recurrence relation that applies to sequentially reacting substances [12]. The impulse response can be written directly in terms of the single reactant impulse response, Eq. (16):

$$h(x,t,K_r,K_a) = \frac{K_d}{K_r-K_a}\left[h(x,t,K_a) - h(x,t,K_r)\right] \qquad (21)$$

where K_a, K_r, and K_d are the reaeration, BOD removal, and deoxygenation rates and the notation $h(x,t,K_a)$ indicates that Eq. (16) is to be evaluated with $K = K_a$. Applying Eq. (13) to this impulse response yields:

$$E\{D(x,t)\} = \frac{K_d}{K_r-K_a}\left[\, E\{c(x,t,K_a)\} - E\{c(x,t,K_r)\}\right] \qquad (22)$$

where D(x,t) is DO deficit concentration and the notation again indicates the reaction rate to be used in the evaluation of Eq. (18). The variance is found from Eq. (14) using Eq. (17) to evaluate the integrals. The result is:

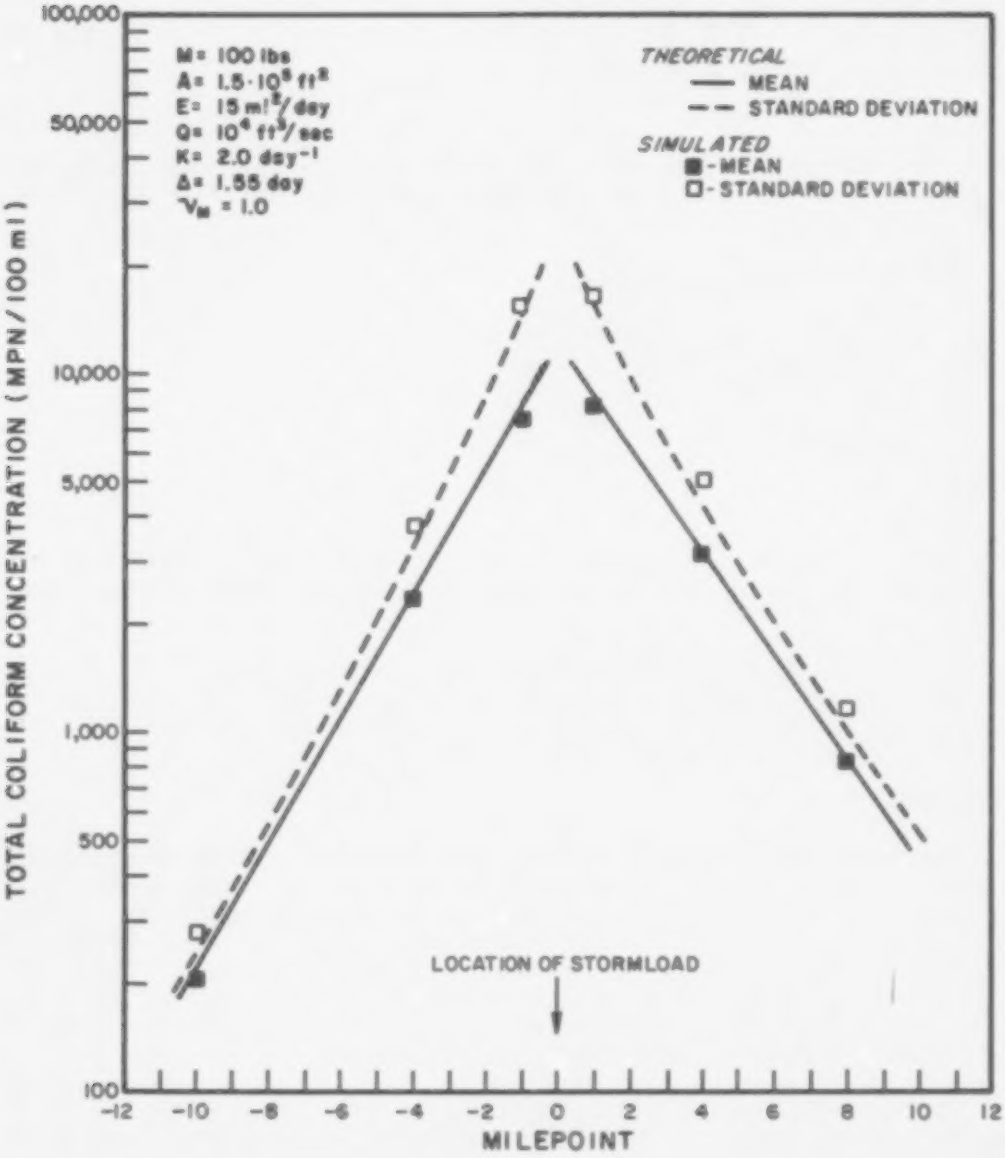


Figure 8: Comparison of theoretical and simulated estuary response for coliform. Receiving water parameters are characteristic of the Hudson River estuary.

$$V\{D(x,t)\} = \frac{M^2(1+\nu_M^2)}{2\pi EA^2\Delta} \left(\frac{K_d}{K_r-K_a}\right)^2 e^p \{K_o(m_a p)$$

$$- 2\, K_o(p\sqrt{1+2\eta_a+2\eta_r}) + K_o\,(m_r p)\} \tag{23}$$

where the subscript notation indicates which reaction rate to use in evaluating the dimensionless expressions: $\eta = KE/U^2$; $m = \sqrt{1+4\eta}$.

The normalized variance:

$$\nu^2\{D(x,t)\} = \frac{\Delta(1+\nu_M^2)U^2}{2\pi E} \tag{24}$$

$$\frac{K_o(m_a p) - 2\, K_o(p\sqrt{1+2\eta_a+2\eta_r}) + K_o(m_r p)}{m_a^{-2}\, \exp(-|p|m_a)\ -2m_a^{-1}m_r^{-1}\, \exp[1-|\frac{p}{2}|(m_a+m_r)]+m_r^{-2}]\exp\,(-|p|m_r)}$$

is symmetric about the point of discharge. And unlike the BOD normalized variance the DO deficit normalized variance at the origin:

$$\nu^2\{D(0,t)\} = \frac{\Delta(1+\nu_M^2)U^2}{2\pi E}\ \frac{\ln\{(1+2\eta_a+2\eta_r)/(m_a m_r)\}}{m_a^{-2} - 2m_a^{-1}m_r^{-1} + m_r^{-2}} \tag{25}$$

is finite.

The mean deficit and normalized variance corresponding to the BOD example in Fig. 5 are shown in Fig. 9. The marked asymmetry of the deficit solution is in marked contrast to the symmetrical normalized deficit variance which is seen to be everywhere smaller than the normalized BOD variance. This is intuititionaly reasonable since there is a kinetic lag time during which BOD is converted to DOD and mixing effects can exert their influence for that much longer.

The normalized deficit variance at the origin can be used to investigate the effect of BOD decay rate and transport. Fig. 10 is a contour plot of $\nu^2\{D(0,t)\}/K_a\Delta(1+\nu_M^2)$ as a function of the ratio of reaeration to deoxygenation coefficient K_a/K_d, and estuary number K_aE/U^2. For highly reactive BOD K_a/K_d is small, the kinetic time lag is reduced, and deficit normalized variance increases. Transport effects are analogous to the single reactant case with more marked increases for $K_aE/U^2<1.0$.

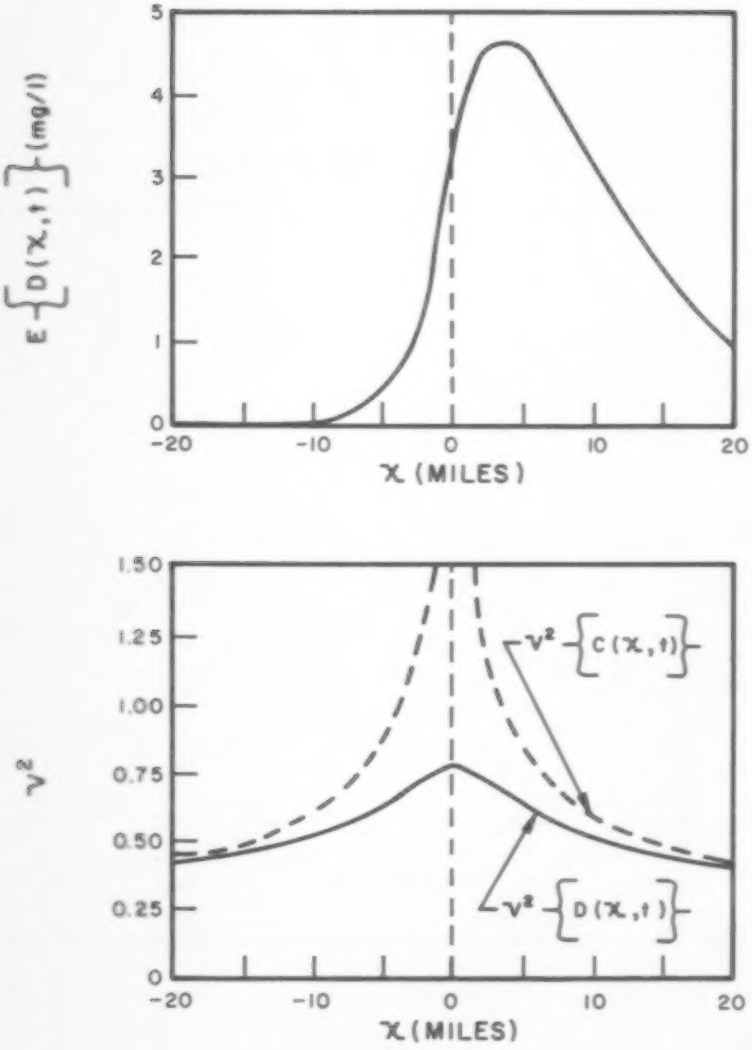


Figure 9: Computed mean and normalized variance for Dissolved Oxygen Deficit response of an hypothetical estuary. Parameters as in Figure 5 with $K_a = 0.15$ day^{-1}

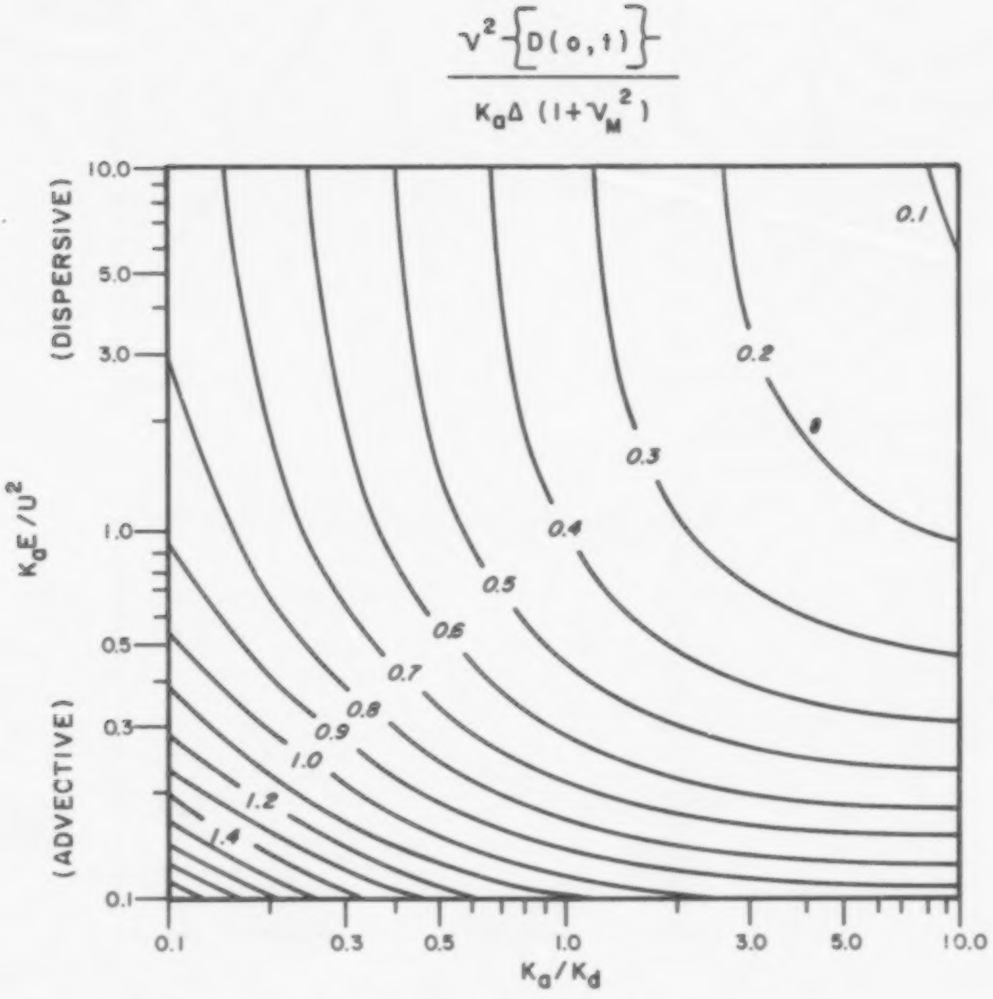


Figure 10: Contour plot of dimensionless normalized Dissolved Oxygen Deficit versus reaeration rate reaction rate ratio and estuary number.

8. Conclusions

The methods presented in this paper can be used to evaluate the statistical behavior of receiving water concentrations that result from runoff events. Certain general results are applicable to a large class of receiving water types. More specialized results apply to receiving waters that can be idealized as a one dimensional rectangular advective-dispersive system for which the runoff flow is not a significant component of the advection. These methods are especially useful for screening many alternative treatment schemes that abate runoff mass discharges. Their effect on the mean and variance of the receiving water can be calculated without costly and time-consuming simulations.

These methods have been extended for the analysis of complex receiving water geometries and multiple discharges. They will be presented in future publications.

9. Acknowledgements

These methods are the results of various research and project related investigations carried out by Hydroscience, Inc. in the past ten years. The initial ideas were developed during conversations with Robert V. Thomann. The contributions of Eugene D. Driscoll, Mitchell Small, Donald J. O'Connor, James Fitzpatrick, John A. Mueller, and Charles Dujardin are specifically acknowledged. The contributions by Dennis Athayde of the Environmental Protection Agency is also gratefully acknowledged.

REFERENCES

1. Hamming, R.W., Numerical Methods for Scientists and Engineers. McGraw-Hill, Inc., 1973, p. 1.

2. Hydroscience, Inc., "A Statistical Method for Assessment of Urban Stormwater" Office of Water Planning and Standards, EPA 440/3-79-D23, Environmental Protection Agency, Washington, D.C., 1979.

3. Cox, D.R. and Lewis, P.A.W., The Statistical Analysis of Series of Events, Methuen & Co., Ltd., London, 1966. p 7-11.

4. Wine, R.L., Statistics for Scientists and Engineers, Prentice-Hall, Englewood Cliffs, N.J., p. 551-552.

5. Mueller, J.A., Anderson, A.R., "Combined Sewer Overflow Quality From Treatment Plant Data", J. Water Poll. Control Fed., Vol. 51., May 1979, p. 958.

6. Rice, S.O., "Mathematical Analysis of Random Noise", in *Selected Papers on Noise and Stochastic Procedures*, ed. N. Wax., Dover Publications, N.Y., 1954, pp. 133-294.

7. Papoulis, A., *Probability, Random Variables, and Stochastic Processes*, McGraw-Hill, N.Y., p. 359, 560-570.

8. O'Connor, D.J., "Analysis of Diffusion Data of the Delaware River Model." Int. J. Air Water Pollution, Vol. 7 pp. 1073-1089, 1963.

9. Erdelyi, A., *Tables of Integral Transforms*, McGraw-Hill, N.Y., 1954, p. 146.

10. Abramowitz, M., Stegun, I.A., *Handbook of Mathematical Functions*, National Bureau of Standards, Applied Mathematics Series 55, Superintendent of Documents, U.S. Government Printing Office, Washington, D.C., 1954, p. 417-429.

11. Di Toro, D.M., O'Connor, D.J., "The Distribution of Dissolved Oxygen in a Stream with Time Varying Velocity", Water Resources Research, Vol. 4, No. 3, June 1968, pp. 639-646.

12. Di Toro, D.M., "Recurrence Relations for First Order Sequential Reactions in Natural Waters", Water Resources Research, Vol. 8, No. 1, Feb. 1972, pp. 50-57.

APPENDIX I

NOTATION

A = cross sectional area

c_i = concentration of i^{th} runoff event

$c(x,t)$ = concentration at location x and time t

d_i = duration of i^{th} runoff event

D = $E\{d_i\}$; mean runoff duration

$D(x,t)$ = dissolved oxygen deficit at location x and time t

E = dispersion coefficient

$E\{\ \}$ = expectation of the quantity

$h(x,t,d_i,q_i)$ = receiving water response to unit ($w_i = 1$) pulse runoff mass discharge at $x = 0$ and centered at $t = 0$, of duration d_i, flow rate q_i.

$h(x,t)$ = receiving water response to unit impulse of mass at $x = 0$, $t = 0$

i_i = rainfall intensity of i^{th} event

K = first order reaction rate

K_r = BOD removal rate

K_a = reaeration rate

K_d = BOD deoxygenation rate

$K_\nu(Z)$ = modified Bessel function of order ν, argument Z

$K_o(Z)$ = modified Bessel function of order zero, argument Z

m_i = mass discharged at i^{th} runoff event

M = mean mass discharged = $E\{m_i\}$

m = $\sqrt{1 + 4\eta}$

m_a = $\sqrt{1 + 4\eta_a}$

m_r = $\sqrt{1 + 4\eta_r}$

p = Ux/E; dimensionless distance

q_i = flow rate of i^{th} runoff event

$r(d_i,t)$ = unit pulse function of duration d_i centered at $t = 0$

t_i = midpoint time of occurrence of the i^{th} event

t = time

U = advective velocity

v_i = volume of the i^{th} runoff event

$V\{\ \}$ = variance of bracketed quantity

w_i = $c_i q_i$; mass discharge rate of the i^{th} runoff event

$W(t)$ = mass discharge rate into advective dispersive system at $x = 0$

x = longitudinal distance

δ_i = $t_{i+1} - t_i$; midpoint interval between runoff events i+1 and i

Δ = $E\{\delta_i\}$; mean midpoint interval

η = KE/U^2; estuary number

η_a = $K_a E/U^2$

η_r = $K_r E/U^2$

ν_δ = $V^{1/2}\{\delta_i\}/\Delta$; coefficient of variation of midpoint interval δ_i

ν_d = $V^{1/2}\{d_i\}/D$; coefficient of variation of duration d_i

ν_i = $V^{1/2}\{i_i\}/E\{i_i\}$; coefficient of variation of intensity i_i

ν_M = $V^{1/2}\{m_i\}/E\{m_i\}$; coefficient of variation of runoff mass, m_i

ν_v = $V^{1/2}\{v_i\}/E\{v_i\}$; coefficient of variation of volume, v_i

$\nu^2\{c(x,t)\} = V\{c(x,t)\}/E^2\{c(x,t)\}$; normalized variance of concentration $c(x,t)$

$\nu^2\{D(x,t)\} = V\{D(x,t)\}/E^2\{D(x,t)\}$; normalized variance of dissolved oxygen deficit

$\rho_\delta(k)$= autocorrelation function of δ_i at lag k; eq. (6)

$\rho_d(k)$ = autocorrelation function of d_i at lag k; eq. (8)

$\rho_i(k)$, $\rho_v(k)$ = autocorrelation function of i_i and v_i at lag k

$\rho_{d\delta}$ = cross correlation coefficient between d_i and δ_i; eq. (7)

$\rho_{q\delta}$, $\rho_{w\delta}$ = cross correlation coefficients between q_i, w_i and δ, respectively

σ_δ = $V^{1/2}\{\delta_i\}$; standard deviation of δ_i

σ_d = $V^{1/2}\{d_i\}$; standard deviation of d_i.

CONTINUOUS RECEIVING WATER QUALITY MODELING FOR URBAN STORMWATER MANAGEMENT

by

Miguel A. Medina, Jr.
Department of Civil Engineering
Duke University
Durham, North Carolina 27706

ABSTRACT

A simplified continuous receiving water quality model has been developed as a planning guide to permit preliminary screening of areawide wastewater treatment strategies. The model simulates the hypothetical response of the stream or tidal river system to the separate and combined effects of waste inputs from: 1) upstream sources, 2) dry weather urban sources, and 3) wet weather urban sources. The total hours of runoff-producing rainfall throughout a year are separated into storm events by defining a minimum interevent time. For a given storm event, the runoff and pollutant loads are summed and critical dissolved oxygen concentrations are estimated as a function of several hydrodynamic and biochemical parameters. Alternative control strategies are evaluated in terms of relative impacts by determining the probability of occurrence of water quality violations. Model output includes the downstream dissolved oxygen sag curves computed per each event, and the dissolved oxygen profile computed at a user-specified location downstream for all simulated events. An application to the Des Moines River at Des Moines, Iowa, is presented.

INTRODUCTION

In a 1.67 square mile (433 hectare) urban watershed in Durham, North Carolina, it was found that the dissolved oxygen content of the receiving watercourse was independent of the degree of municipal waste treatment beyond secondary during storm flows

(3). Approximately one-half of the stream miles in the United States are water quality limited and 30 percent of these stream segments are considered polluted to a certain degree with urban stormwater runoff (5). The implication is that, generally, secondary treatment of dry-weather wastewater flows is insufficient to meet desired receiving water quality standards: therefore, control of runoff pollution must be considered in areawide wastewater management plans and abatement programs. The results of a nationwide assessment of costs and related water quality impacts derived from non-point sources (7) were, among others, that:

- wet-weather flows represent at least 50 percent of the total wastewater flow from urban areas;
- a generalized optimization model, assuming linear costs, predicted primary type facilities are preferable only up to a 10 percent level of BOD removal for wet-weather flows, with a secondary type facility preferable for higher levels of control; and
- on a national average basis using BOD removal as the effectiveness parameter, approximately 39 percent of the combined sewer problem and 10 percent of the other wet-weather flows should be controlled before initiating tertiary treatment of point sources.

The study also confirmed that gross inadequacies exist in our present data base and conclusions are highly sensitive to simplifying assumptions necessary for successful simulation of complex physical processes occurring throughout our watersheds. Nevertheless, mathematical models are needed to predict variable responses to stochastic hydrologic phenomena.

The 208 planning effort (Section 208, PL 92-500) established the need for various levels of urban water management analysis (5) to permit preliminary screening of municipal treatment alternatives. Four distinct levels of evaluation techniques, ranging from simple to complex procedures which can be integrated with one another, are summarized in Table 1. The first three levels essentially represent various degrees of planning detail with models running on hourly time steps for long simulation periods (years). Mathematical complexity and data requirements are kept at a minimum. At the third level, the methods used to generate storm runoff flows and pollutant mass rates or concentrations depend upon the continuous hydrologic simulation model selected by the user. Such techniques are well documented for two models which may be applied to both urban and non-urban watershed: 1) the Hydrologic Engineering Center's Storage, Treatment, Overflow, Runoff Model (STORM), and 2) the U. S. Environmental Protection Agency's Storm Water Management Model (SWMM), continuous version of Runoff Block (17,9). The urban runoff flows and associated concentrations and mass rates, derived from storm events over the drainage area of

TABLE 1. LEVELS OF URBAN WATER MANAGEMENT ANALYSIS DEVELOPED BY EPA RESEARCH & DEVELOPMENT PROGRAMS

Level I: a desktop calculator, statistical analysis procedure, no electronic digital computer required.

- University of Florida Methodology - permits the user to estimate the quantity and quality of urban runoff in the combined, storm and unsewered portions of each urban area.
- Hydroscience, Inc., Methodology - use of a stormwater simulator and an analytical method based on probability distribution functions and statistical properties of rainfall, runoff, treatment and receiving water impact.

Level II: a simplified continuous simulation model for planning and preliminary sizing of facilities, developed by Metcalf & Eddy, Inc.; or the computerized optimization version of University of Florida Methodology described above.

Level III: a refined continuous simulation model approach. Continuous hydrologic simulation models (e.g., STORM or continuous SWMM) which generate urban runoff hydrographs and pollutographs are followed by continuous receiving water impact analyses (Level III-Receiving model).

- Continuous SWMM - University of Florida
- STORM - Corps of Engineers by Water Resources Engineers, Inc.

Level III-Receiving -- Duke University

Level IV: a sophisticated single event simulation model, e.g., EPA SWMM developed by Metcalf & Eddy, Inc., University of Florida, and Water Resources Engineers, Inc.

interest, represent concurrent time series which are read by Level III-Receiving through the standard card reader devices, or from peripheral storage units (disk/tape/drum). Since the program has the built-in capability of accessing a user-created data set, any continuous hydrologic and water quality model may be considered (15). Detailed models such as the single-event version of SWMM typically use short time steps (minutes) and short simulation times (hours). Their data requirements are usually substantial.

MODEL OVERVIEW

Since it is impractical, if not impossible, to meet stream standards all of the time it appears desirable to express allowable limits for various pollutants on a probabilistic or frequency basis. The approach guiding the development of Level III-Receiving was that the cost-effectiveness of various treatment alternatives can only be determined realistically by a continuous analysis of the frequency of violation of water quality standards. Such an analysis provides an approximation of system responses to proposed treatment measures and must never be interpreted as other than a technique in the selection of the best control strategy.

The problem of specific interest is to assess the separate and combined effects of the major urban sources of water pollution upon river quality. The relative impact of wastewater sources is appraised by estimating their effect on dissolved oxygen (DO) concentrations downstream from the urban area. It is of further interest to distinguish clearly between the relative pollutional impacts of storm sewer discharges and combined sewer overflows. An abstraction of the urban drainage system is presented in Fig. 1. The urban community served by a separate sewer system will convey stormwater runoff and municipal sewage through parallel conduits. The BOD concentration of the storm sewer runoff is mixed with the dry-weather flow (DWF) and accumulated sewer solids. An interceptor carries the sanitary design flow to the municipal sewage treatment plant. The combined sewer overflow is either given treatment or allowed to discharge directly to the receiving water. Since complete mixing is assumed, the BOD concentrations of the combined sewer overflow (Q_c) and the flow (DWFCMB) intercepted for treatment by the DWF facility are identical. Any desired degree of treatment may be imposed at both the DWF and the wet-weather flow (WWF) treatment plants. The concentration of the combined BOD inputs in the receiving water is given by:

$$BOD_m = \frac{BOD_u Q_u + BOD_d Q_d + BOD_w Q_w}{Q_u + Q_d + Q_w} \quad . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; . \quad (1)$$

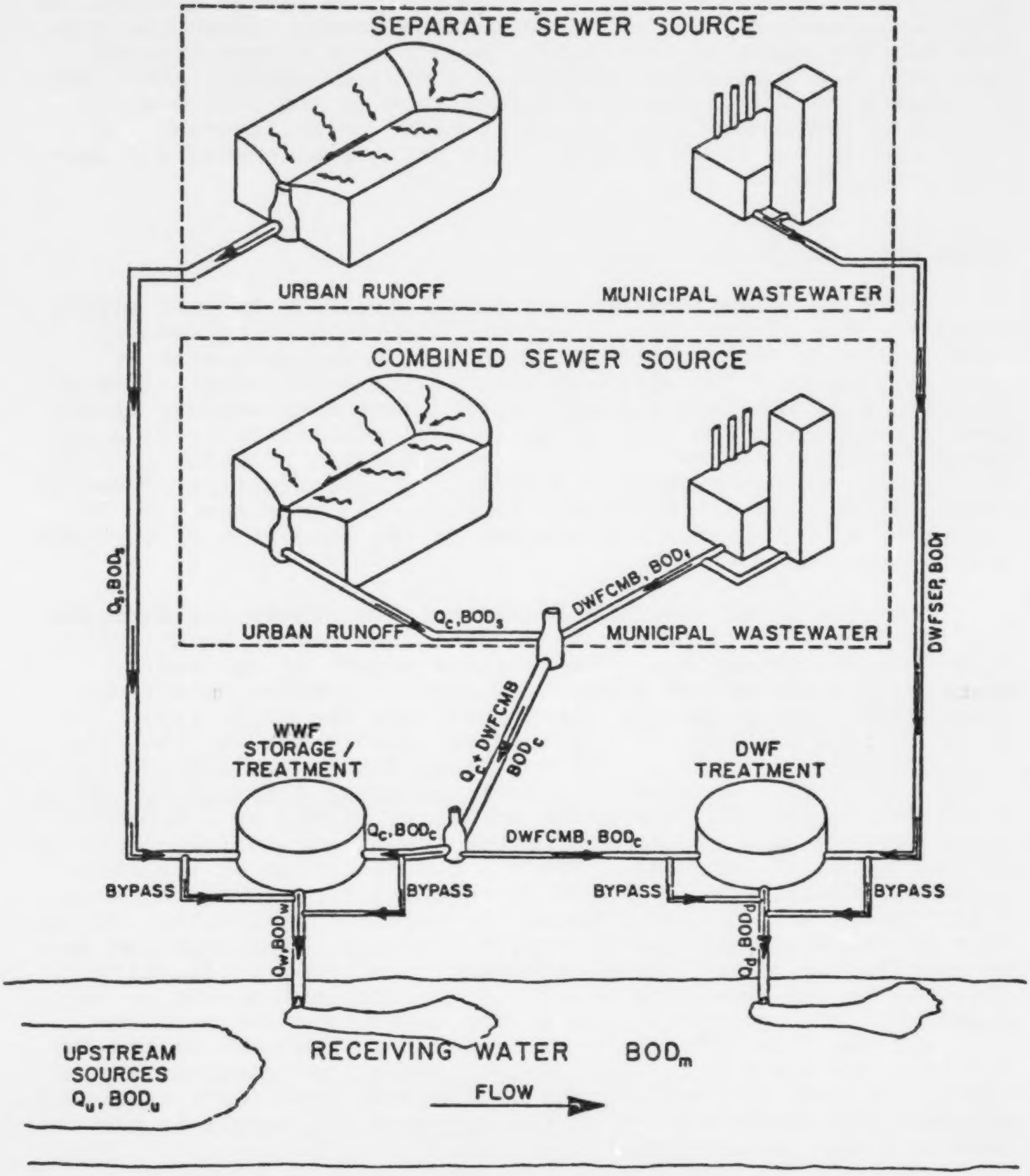


Fig. 1. Abstraction of the
Urban Drainage System

where BOD_m = mixed BOD concentration in receiving water, mg/l,

BOD_u = mixed BOD concentration from sources upstream of urban area, mg/l,

BOD_d = BOD concentration of dry-weather flow treatment plant effluent, mg/l,

BOD_w = BOD concentration of wet-weather flow treatment facility effluent, mg/l,

Q_u = upstream flow, cfs,

Q_d = DWF treated effluent, cfs, and

Q_w = WWF treated effluent, cfs.

The BOD concentrations of the DWF and WWF treated effluents are given by:

$$BOD_d = \frac{[BOD_f \cdot DWFSEP + BOD_c \cdot DWFCMB]\ (1-R_d)}{DWFSEP + DWFCMB} \quad . \ . \ . \ . \ (2)$$

$$BOD_w = \frac{[BOD_s \cdot Q_s + BOD_c \cdot Q_c]\ (1-R_w)}{Q_s + Q_c} \quad . \ . \ . \ . \ . \ . \ . \ . \ . \ (3)$$

where BOD_f = BOD concentration of municipal sewage, mg/l,

BOD_c = mixed BOD concentration in the combined sewer, mg/l,

BOD_s = BOD concentration of urban stormwater runoff, mg/l,

DWFSEP = DWF contribution from separate sewer area, cfs,

DWFCMB = DWF contribution from combined sewer area, cfs,

Q_s = urban runoff carried by the separate storm sewer, cfs,

Q_c = combined sewer flow, cfs,

R_d = fraction removal of BOD achieved by the DWF treatment facility, and

R_w = fraction removal of BOD achieved by the WWF treatment facility.

The initial conditions of BOD in the river are defined by Eq. 1, and the hypothetical impact on the oxygen balance of the receiving stream is estimated by using simplified mathematical modeling approaches. The total hours of runoff-producing rainfall throughout the year are separated into storm events by defining a minimum interevent time. For a given storm event, the runoff and pollutant loads are summed and the critical DO deficit is estimated as a function of several stream parameters: temperature, flow, oxygen concentration, deoxygenation and reaeration rates, longitudinal dispersion, and BOD concentrations. The minimum DO is calculated subsequently and a frequency analysis is performed. Stream velocity is computed as a function of the discharge and the time and distance to each critical deficit point are obtained for each event.

The options used for the simulations include:

1. five inflow combinations of river flow plus:
    a. DWF
    b. DWF + separate flow
    c. DWF + combined flow
    d. separate flow + combined flow
    e. separate flow + combined flow + DWF,
2. four DWF treatment rates (variable),
3. three WWF treatment rates (variable), and
4. three fractions of measured upstream flow.

Item 4 is included as a model option to investigate whether the relative impact of urban stormwater runoff is significant in the upstream portions of river basins. This effect may be simulated by simply reducing the upstream flow to any desired fraction of its actual measured value. Thus, discharge into a dry river bed may be studied.

The impact of urban runoff on the quality of the receiving stream is evaluated in terms of: 1) critical DO frequency curves, i.e., by the percent of time predicted minimum DO levels fall below a specified concentration; and 2) classical DO sag curves for each inflow combination. During periods of wet weather, the program performs computations on an hourly time step. Otherwise, a daily time step is used. An economic evaluation of various control strategies augments the decision-making capabilities of such an approach.

EVENT DEFINITION

The basic approach to define a wet-weather event is to analyze the runoff time series and establish the minimum number of consecutive dry-weather hours (DWH) that separates independent storm events. The independence of these events is not defined in a strictly climatologic sense; it is in fact statistically derived. The DWH refer to periods during which no runoff was produced. If STORM is selected to generate the hydrologic time series, depression storage and evaporation rates must be satisfied before any runoff is predicted and no runoff occurs during periods without measurable precipitation. The continuous version of SWMM allows runoff to decay temporally beyond intervals with zero precipitation input. Therefore, for an identical precipitation time series, runoff events generated by SWMM will generally be of longer duration.

The runoff time series is subjected to autocorrelation analysis. For hydrologic processes, it is practical to estimate the autocorrelation coefficients by an open-series approach (6,18):

$$r_I(k) = \frac{\sum_{i=1}^{n-k} x_i x_{i+k} - \frac{1}{n-k}\left[\sum_{i=1}^{n-k} x_i\right]\left[\sum_{i=k+1}^{n} x_i\right]}{\left[\sum_{i=1}^{n-k} x_i^2 - \frac{1}{n-k}\left(\sum_{i=1}^{n-k} x_i\right)^2\right]^{0.5}\left[\sum_{i=k+1}^{n} x_i^2 - \frac{1}{n-k}\left(\sum_{i=k+1}^{n} x_i\right)^2\right]^{0.5}} \quad . . (4)$$

where $r_I(k)$ = sample estimate of lag-k autocorrelation coefficient for hydrologic process I,

x_i = discrete data series (observations) of hydrologic process I, for $i = 1,2,...,n$,

n = total number of data points or observations, and

k = number of hourly lags.

The tolerance limits for a normal random time series which is circular and of lag 1 is given by (1):

$$\text{TL}\ [r_I(1)] = \frac{-1 \pm t_\alpha \sqrt{n-2}}{n-1} \quad . . . . (5)$$

where t_α = standardized normal variate corresponding to probability level $1 - \alpha$.

A circular time series is defined as a series where the last value is followed by the first so that the time series repeats itself. Eq. 5 has been extended for use with an open series, for the general lag case (18). At a 95 percent probability level, the tolerance limits are given by:

$$\text{TL}\ [r_I(k)] = \frac{-1 \pm 1.645\ \sqrt{n-k-1}}{n-k} \quad . . . . (6)$$

A plot of the serial correlation coefficients, $r(k)$, against the number of lags, k, is called a correlogram. The technique of autocorrelation analysis is essentially a study of the behavior of the correlogram of the process under investigation (16). The model compares the value of $r(k)$ obtained from Eq. 4 with TL $[r_I(k)]$, computed by Eq. 6, for the corresponding number of hourly lags k. The minimum interevent time (MIT) which separates independent wet-weather events is defined as the minimum value of k for which $r(k)$ is not significantly different from zero at a 95 percent probability level.

EFFECT ON RECEIVING WATERS

A simplified mathematical modeling approach is used in which critical deficits and resulting minimum DO concentrations are determined for a large number of waste input combinations, treatment schemes, and receiving water conditions. The development of a detailed and sophisticated model is not justified for the problem context: to provide adequate information on the relative effectiveness of various pollutant control strategies in achieving selected water quality standards. The basic theory of mathematical modeling of one-dimensional bodies of water is applied for the spectrum of natural systems from freshwater streams to tidal rivers. The approach is particularly advantageous for a limited data base on natural system geometry, hydrodynamic variables, and water quality. Assumptions typical of models limited for preliminary planning are made (10):

(1) Temporal steady-state conditions prevail, where all system parameters and inputs are constant with respect to time, however, a relatively short time step (1 hour) is used for simulation.

(2) Natural system parameters (such as flow, velocity, hydraulic depth, deoxygenation and reaeration rates, and longitudinal dispersion) are spatially constant along the flow axis throughout each time step.

(3) All waste inflows to the receiving body of water occur at one point.

(4) The effects of various natural biological processes (algal photosynthesis and respiration, benthal stabilization) are incorporated into a background quality which is reflected by DO deficit (if none, by saturation) upstream from the waste inflow point. Any benthic buildup is incorporated into the BOD decay rate.

(5) Waste treatment facilities operate at constant efficiencies, independent of hydraulic and organic

loadings, for the entire period of simulation.

Oxygen Balance of Polluted Streams and Tidal Rivers

In view of the modeling objectives, pollutant transport processes in these systems are approximated by the steady-state, one-dimensional version of the classical convective diffusion equation:

$$o = \frac{\partial}{\partial x} \left[E \frac{\partial C}{\partial x} - UC \right] \pm \Sigma S \quad . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; (7)$$

where C = concentration of water quality parameter (pollutant), M/L^3,

$-E \frac{\partial C}{\partial x}$ = mass flux due to longitudinal dispersion along the flow axis, the x direction, M/L^2T,

UC = mass flux due to advection by the fluid containing the mass of pollutant, M/L^2T,

S = sources or sinks of the substance C, M/L^3T,

U = flow velocity, L/T, and

E = longitudinal dispersion coefficient, L^2/T.

The equation assumes no diffusion of pollutants through the body of water boundaries (other than what may be included in the source-sink term) and is best suited to predict concentrations relatively far downstream from the point of waste injection. Since critical DO deficits usually occur some distance downstream from the waste source, Eq. 7 is adequate for such predictions. The main sources of dissolved oxygen in stream systems are atmospheric reaeration and oxygen production by photosynthesis. The major sinks include carbonaceous oxygen demand (CBOD), nitrogenous oxygen demand (NBOD), benthal demand, and respiration of aquatic plants. In the model, the effects of photosynthesis, respiration and benthal demand are assumed to be incorporated into the measured upstream DO deficit. Field measurements of organic and ammonia nitrogen present in all wastewater inputs to receiving waters are seldom available, much less during runoff events. Thus, nitrogenous oxygen demand is currently neglected. Since it is desired to solve for the DO deficit and

$$\frac{\partial C}{\partial x} = - \frac{\partial D}{\partial x} \quad . \; (8)$$

Eq. 7 becomes:

$$E \frac{d^2D}{dx^2} - U \frac{dD}{dx} + K_1 L_o e^{mx} - K_2 D = 0 \quad . . . . (9)$$

where

$$m = \frac{U}{2E}\left[1 - \sqrt{1 + \frac{4K_1E}{U^2}}\right] \quad \text{for } x \geq o$$

D = DO deficit, mg/l,

E = longitudinal dispersion coefficient, ft^2/hour,

U = freshwater stream or tidal river velocity, ft/hour,

K_2 = atmospheric reaeration coefficient, $hours^{-1}$,

K_1 = deoxygenation constant of carbonaceous BOD, $hours^{-1}$,

L_o = remaining carbonaceous BOD concentration at $x = o$, mg/l,

Critical Deficit and DO Levels

The solution to Eq. 9 as a function of time since release is given by

$$D = \frac{L_oK_1}{K_2 - K_1}\left[e^{jt} - \frac{s_1}{s_2} e^{gt}\right] + D_o e^{gt} \quad . . . . (10)$$

where D_o = initial DO deficit, mg/l,

$t = \frac{x}{U}$ = translational time, hours,

$j = \frac{U^2}{2E}\left[1 - \sqrt{1 + \frac{4K_1E}{U^2}}\right]$, $hours^{-1}$,

$g = \frac{U^2}{2E}\left[1 - \sqrt{1 + \frac{4K_2E}{U^2}}\right]$, $hours^{-1}$,

$s_1 = \sqrt{1 + \frac{4K_1E}{U^2}}$, dimensionless, and

$s_2 = \sqrt{1 + \frac{4K_2E}{U^2}}$, dimensionless.

The time at which the critical (maximum) deficit occurs is obtained by taking the partial derivative of Eq. 10 with respect to time, setting the resulting equation to zero and solving for time. After simplification and convenient substitutions,

$$t_c = \frac{1}{(j - g)} \ln \left[\frac{g}{j} \left(\frac{s_1}{s_2} - fR_o + R_o \right) \right] \qquad \text{. . . . . (11)}$$

where t_c = elapsed time at which the critical deficit occurs, hours,

f = self-purification ratio

= K_2/K_1, and

R_o = ratio of the initial DO deficit, D_o, to the initial BOD, L_o, dimensionless.

Finally, the critical deficit is found by substituting the value of t_c, given by Eq. 11 into Eq. 10:

$$D_c = \frac{L_o K_1}{K_2 - K_1} \left[e^{j t_c} - \frac{s_1}{s_2} e^{g t_c} \right] + D_o e^{g t_c} \qquad \text{. . . . . . (12)}$$

where D_c = critical (maximum) deficit, mg/l.

The minimum DO level is calculated as

$$C_{min} = C_s - D_c \qquad \text{. . . . . (13)}$$

where C_{min} = concentration of DO at maximum deficit, mg/l, and

C_s = saturation concentration of DO, mg/l.

The saturation concentration is determined from the regression relationship (2),

$$C_s = 14.652 - 0.41022\, T + 0.0079910\, T^2 - 0.000077774\, T^3 \qquad \text{. . . (14)}$$

where T = water temperature, °C.

For freshwater streams, the advective flux is significantly larger than the mass flux due to longitudinal dispersion. In a tidal river, the advective and dispersive fluxes are both significant. However, it should be noted that even for large, deep rivers, the effect of dispersion generally cannot be neglected when the wastewater inputs are time variable (13,14). If the model user wishes to neglect the dispersive flux, the classical Streeter-Phelps formulations are then applied.

The Langbein and Durum (11) equation for the prediction of the reaeration coefficient was chosen because it is most closely related to subsequent procedures applied to obtain stream velocity and depth:

$$K_2 = 2.303 \left[3.3 \frac{U}{H^{1.33}} \right] \quad \ldots\ldots\ldots\ldots\ldots \quad (15)$$

where K_2 = reaeration coefficient at 20°C, day^{-1},

U = stream velocity, ft/sec, and

H = stream depth, ft.

The problem lies in obtaining values of U and H, since the streamflow varies with time. In the absence of measurements, or if the data cannot be obtained in an expedient manner (as in the ensuing application to the Des Moines River), an approximation can be made (12) which uses strong correlations between velocity versus flow and depth versus flow, namely:

$$U = \alpha_1 Q_2^{\alpha} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (16)$$

$$H = \beta_1 Q_2^{\beta} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (17)$$

where Q = streamflow, cfs, and

$\alpha_1, \alpha_2, \beta_1, \beta_2$ = regression coefficients.

The model user also has the option of computing the deoxygenation coefficient, K_1, as a power function of water depth (10):

$$K_1 = \gamma_1 H_2^{\gamma} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (18)$$

where γ_1, γ_2 = regression coefficients.

Temperature corrections are applied to carbonaceous BOD, deoxygenation and reaeration rate coefficients.

PROGRAM OPERATION

Model subprogram interactions are shown in Figure 2. The main program (hereafter referred to as subroutine MAIN) provides overall control and includes in its entirety the wet-weather flow model ((WWFM). The first input data card to subroutine MAIN allows the user to select which subprograms will be executed during simulation. The WWFM is a mathematical abstraction of the physical system depicted earlier in Figure 1. Stormwater runoff flows and pollutant loads or concentrations generated by an urban, continuous hydrologic simulation model (e.g., STORM or SWMM) are read either directly from card input or from peripheral devices (tape/disk/drum) depending on the machine configuration. The WWFM in subroutine MAIN simulates the conveyance system, including mixing in combined sewers of wet-weather and dry-weather pollutants during periods of runoff; the pollutant removal efficiency of various treatment schemes; mixing of the various pollutant inflow combinations with upstream sources in the receiving waters to determine initial conditions of BOD, DO, streamflow and other parameters; and computes the oxygen balance of the polluted waters downstream from the waste sources. The procedure continues for each independent storm event as defined by the minimum interevent time (MIT). Pollutant loadings and receiving water quality conditions are averaged over *each event's total duration*, which includes wet-weather and dry-weather hours. The spatial distribution of dissolved oxygen concentrations along the flow axis is computed for each event for a distance downstream chosen by the model user. Frequency analyses are performed on either the resultant DO concentrations at a specified location downstream, or critical (minimum) DO concentrations as predicted by the model for the entire period of simulation.

Subroutine CORREL subjects the hydrologic time series (either rainfall or runoff) to autocorrelation analysis. It automatically defines the MIT used in the WWFM to separate wet-weather events. The methodology has been discussed previously. This subroutine may be executed independently of the WWFM, but may be accessed only through subroutine MAIN. Subroutine MGRAPH, a single and multiple-curve plotting subprogram, is called by CORREL to display the correlogram of the time series. MGRAPH, in turn, calls subroutine ROUND to set the appropriate scale on the coordinate axes from examination of minimum and maximum values to be plotted.

Subroutine DWFM performs the same functions as the WWFM, *during periods of no urban runoff*. Thus, a model assumption is *that no combined sewer overflows* occur. Therefore, there is no "first-flush" effect, and there are no storm events over which to average pollutant loads. It may be executed independently of all other subroutines, except MAIN. Subroutine PLOT is called by the WWFM (contained in subroutine MAIN) and also by subroutine DWFM to display frequency *histograms* of receiving water DO concentrations, optionally. Likewise, subroutine MGRAPH is

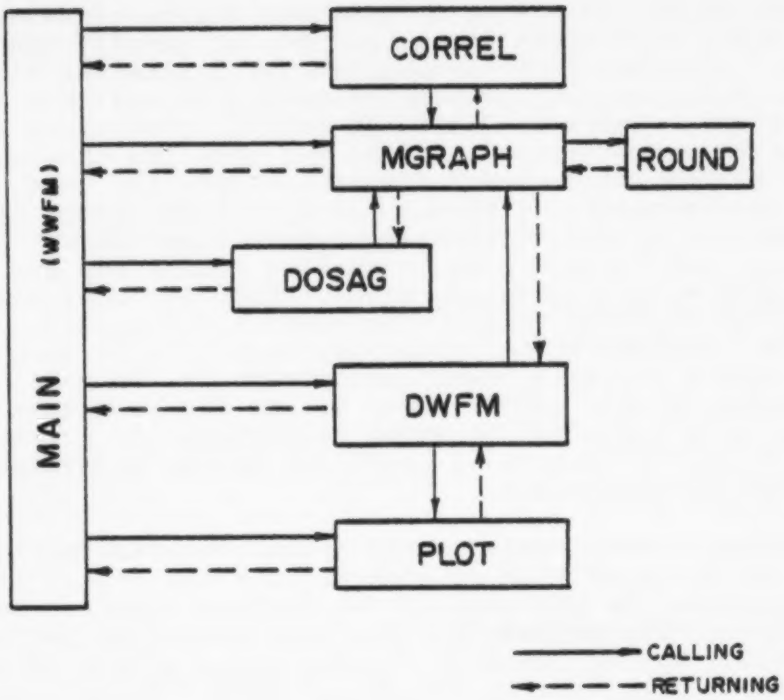


Figure 2. Level III-Receiving Subprograms

called by both models to plot cumulative, multiple frequency curves of DO concentrations.

Subroutine DOSAG is called from subroutine MAIN to display in tabular form and chronological order: the computed dissolved oxygen concentration at a user-specified location downstream, for each event simulated by the WWFM only, the DWFM only, or both. Thus, the listing may include DO concentrations resulting from wet-weather event pollutant loadings as well as DWF pollutant loadings during periods of no urban runoff. Consequently, the subprogram sorts the values according to date of occurrence in order to produce a composite chronological record. These DO concentrations are read by DOSAG from scratch data sets created automatically by the WWFM and the DWFM. Subroutine MGRAPH is called by subroutine DOSAG to provide a plot corresponding to the tabular listing. The plot ordinate (dissolved oxygen concentration) is scaled according to magnitude; however, the abscissa represents the total number of simulated events, numbered sequentially and not scaled according to their time of occurrence since the beginning of simulation. Each event number is identified according to date in a tabular format which precedes each plot.

APPLICATION TO STUDY AREA

The City of Des Moines, Iowa, is located near the confluence of the Des Moines River and the Raccoon River. It contains approximately 200,000 people out of the total of 288,000 for the metropolitan area (4). The mean annual precipitation is 31.27 inches (795 mm) which is approximately equal to the United States average, and the average value for the State of Iowa. The urban area covers 49,000 acres (19,830 ha) of land which has gently rolling terrain. Most of the area, 45,000 acres (18,211 ha), is served by separate sewers, while 4,000 acres (1619 ha) are served by combined sewers.

Selection of the study area was based primarily on data availability. Davis and Borchardt (4) conducted an extensive sampling program of combined sewer overflows, stormwater discharges, and surface waters in the Metropolitan Des Moines area as part of a combined sewer overflow abatement planning study. The sampling program was conducted from March 1968 to October 1969. Other considerations revolved around the fact that Des Moines, Iowa, is somewhat typical of many urban centers throughout the country:

(1) it has a medium-sized population;
(2) its domestic and industrial dry-weather flows receive secondary treatment;
(3) its wastewaters are discharged into a non-tidal receiving stream; and

(4) the urban area receives a mean annual precipitation equal to the national average.

The autocorrelation function of hourly runoff events for Des Moines, Iowa, is presented in Fig. 3. At a number of hourly lags equal to zero, the correlation of the discrete open series is unity because this point represents the linear dependence of the data series on itself. An essentially zero correlation (within 95 percent tolerance limits) between runoff events first occurs at a lag of approximately 9 hours, defining the minimum interevent time. The physical interpretation is that periods without runoff for at least 9 hours separate uncorrelated, and therefore independent, wet weather events. The hourly urban runoff and associated pollutant loads within each event (including DWF pollutant loads during DWH periods less than nine hours duration) are summed, average conditions are determined, and the model proceeds with the receiving water analysis.

Model calibration was preceded by calibration of the urban runoff BOD_5 loading rates for Des Moines, Iowa, as computed by STORM. The dust and dirt surface loading factors were adjusted to obtain an annual average flow-weighted BOD_5 concentration of 53 mg/l for urban stormwater runoff. The above concentration was the average value determined by the field monitoring program in the separate sewer system. The developed mathematical model, as discussed in the overview, simulates the mixing of stormwater runoff and sanitary sewage in the combined sewer system. The annual average flow-weighted BOD_5 concentration of combined sewer overflows was computed to be 75 mg/l, including the effects of first flush (15). The average value measured in the combined sewer system was reported to be 72 mg/l. Model parameters were adjusted to obtain an adequate fit between calculated and observed profiles of dissolved oxygen. These curves are shown in Figure 4, along with other pertinent information, and correspond to a point 5.6 mi (9.0 km) downstream from the confluence of the Raccoon and Des Moines Rivers.

Based on precipitation records obtained from the Environmental Data Service, the total rainfall that fell over Des Moines, Iowa, during 1968 was 27.59 inches (701 mm). STORM computed a total runoff of 10.28 inches (261 mm) over a watershed area of 49,000 acres (19,600 ha), for an overall urban area runoff coefficient of 0.37. There were 65 days in the year during which rainfall was recorded, from which 58 wet-weather events were defined. The results are presented in the form of minimum DO frequency curves for the wet-weather and dry-weather periods throughout the calendar year.

As stated in the overview, a large number of wastewater inflow and treatment rate combinations were investigated and the

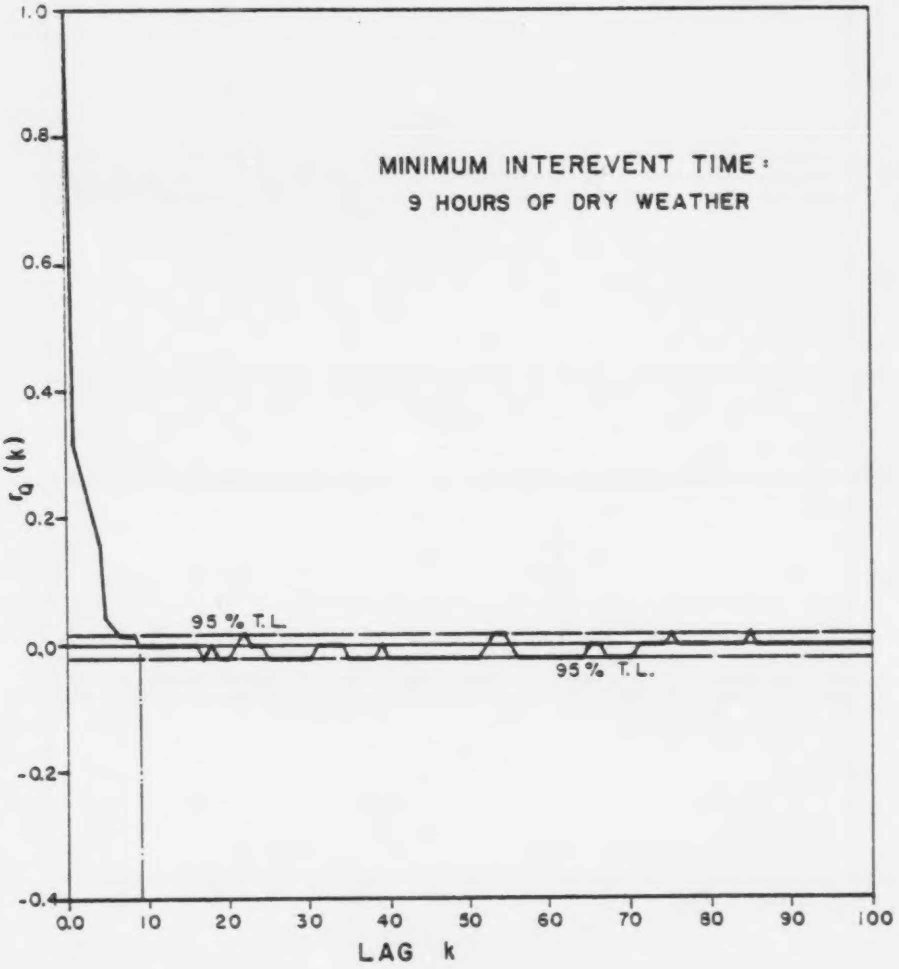


Fig. 3. Autocorrelation Functions of Hourly Urban Runoff for Des Moines, Iowa, 1968.

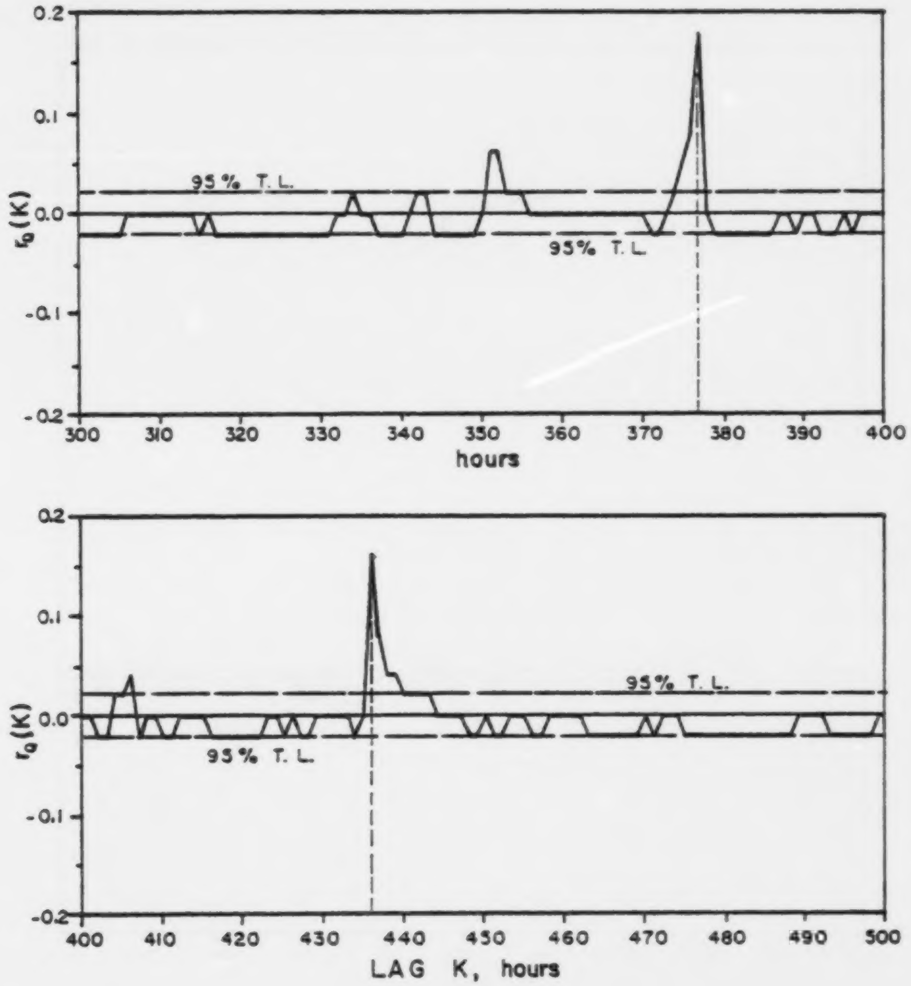


Fig. 3 (continued). Autocorrelation Function of Urban Runoff for Des Moines, Iowa, 1968.

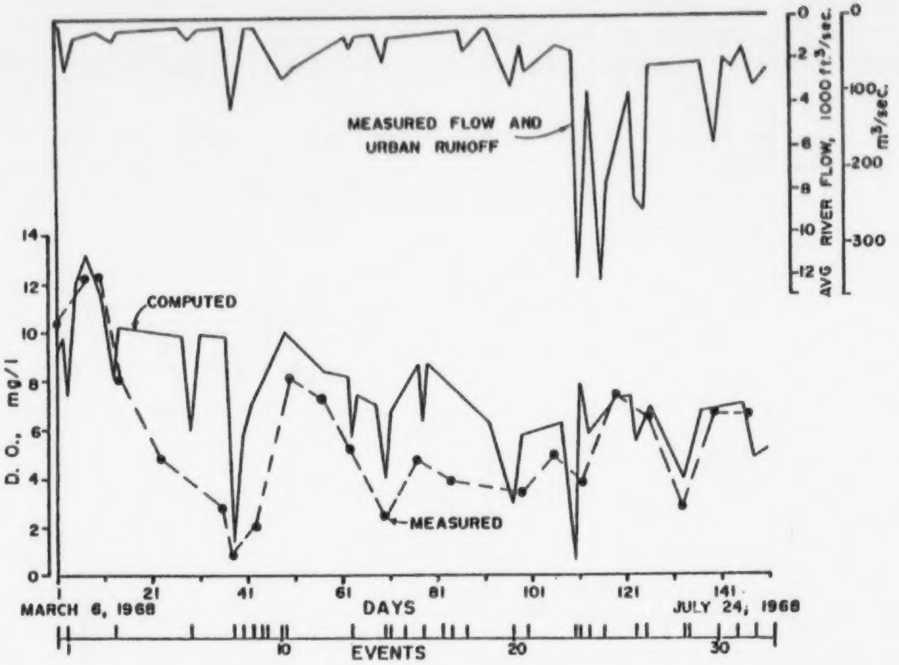


Fig. 4. Application to Des Moines, Iowa. Measured and computed values of DO at 5.6 mi (9.0 km) downstream from confluence of Raccoon and Des Moines Rivers.

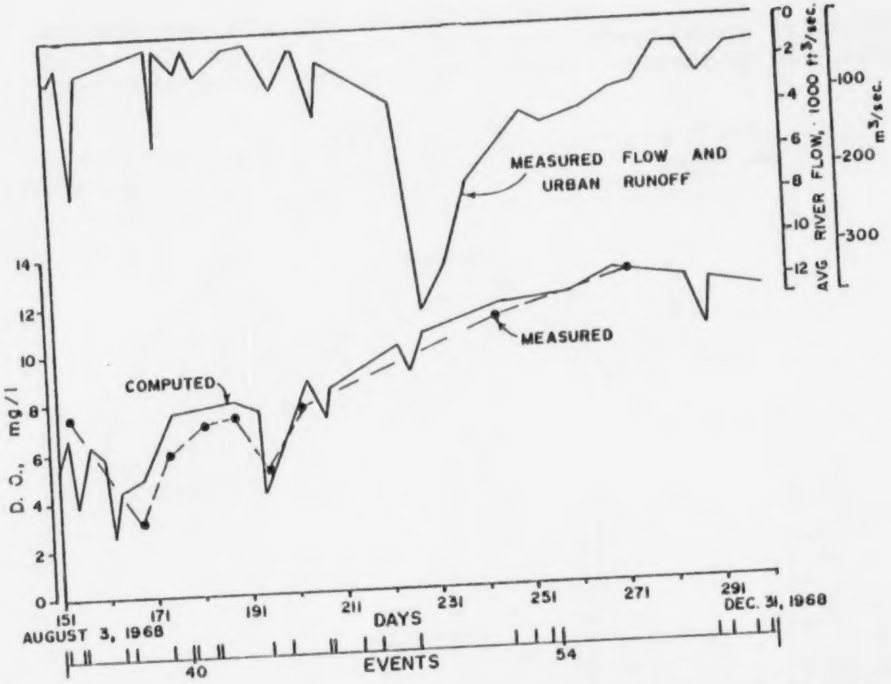


Fig. 4 (continued). Application to Des Moines, Iowa. Measured and computed values of DO at 5.6 mi (9.0 km) downstream from confluence of Raccoon and Des Moines Rivers.

complete results are available elsewhere (15). A representative sample is presented henceforth. It is appropriate to examine the model estimates of critical DO concentration in the Des Moines River, for all waste inputs, for conditions assumed to exist in 1968 during periods of urban runoff: (1) secondary treatment (85 percent BOD removal) of DWF; (2) no stormwater treatment; (3) full river flow (100 percent of measured flow); and (4) combined sewer area equal to 8.16 percent of the total urban area. Fig. 5 represents the minimum DO frequency curves for these conditions. The curves indicate clearly that all combinations including a substantial amount of wet weather flow (WWF) result in a drastic decrease in river minimum DO concentrations. For example, 63 percent of all the wet weather events throughout the year produced conditions in the receiving water that caused minimum DO levels below 4.0 mg/l. Combined sewers contributed WWF from only 8 percent of the total urban area modeled, yet the BOD_c concentration was sufficiently high to inflict an appreciable reduction in DO levels when compared to DWF sources during periods of runoff.

The minimum DO frequency curves in Fig. 6 compare six treatment alternatives to reduce water pollution during periods of urban runoff:

1. 85 percent treatment of DWF and 75 percent treatment (BOD removal) of WWF,
2. 85 percent treatment of DWF and 50 percent treatment of WWF,
3. 85 percent treatment of DWF and 25 percent treatment of WWF,
4. 95 percent treatment of DWF and no treatment of WWF,
5. 85 percent treatment of DWF and no treatment of WWF, and
6. 30 percent treatment and no treatment of urban runoff.

Even option 3, with a small percentage of urban runoff treatment, is superior to all alternatives controlling DWF only. An economic evaluation of these treatment alternatives, on an annual basis, is presented subsequently.

Dry weather was experienced for approximately 300 days throughout 1968. The model was applied to these periods using a daily time step. This modification is certainly justified since conditions are more truly steady-state than during periods of precipitation and subsequent runoff: for example, waste loadings (DWF treatment plant effluent) and river flow do not vary as much during the day. For the dry weather simulation period, upstream river flow was on the average 94 percent of total river flow, ranging from 82 percent to 99.6 percent. The results are shown in Fig. 7. A remarkable 97 percent of the dry weather days exceed a minimum DO concentration of 4.0 mg/l with

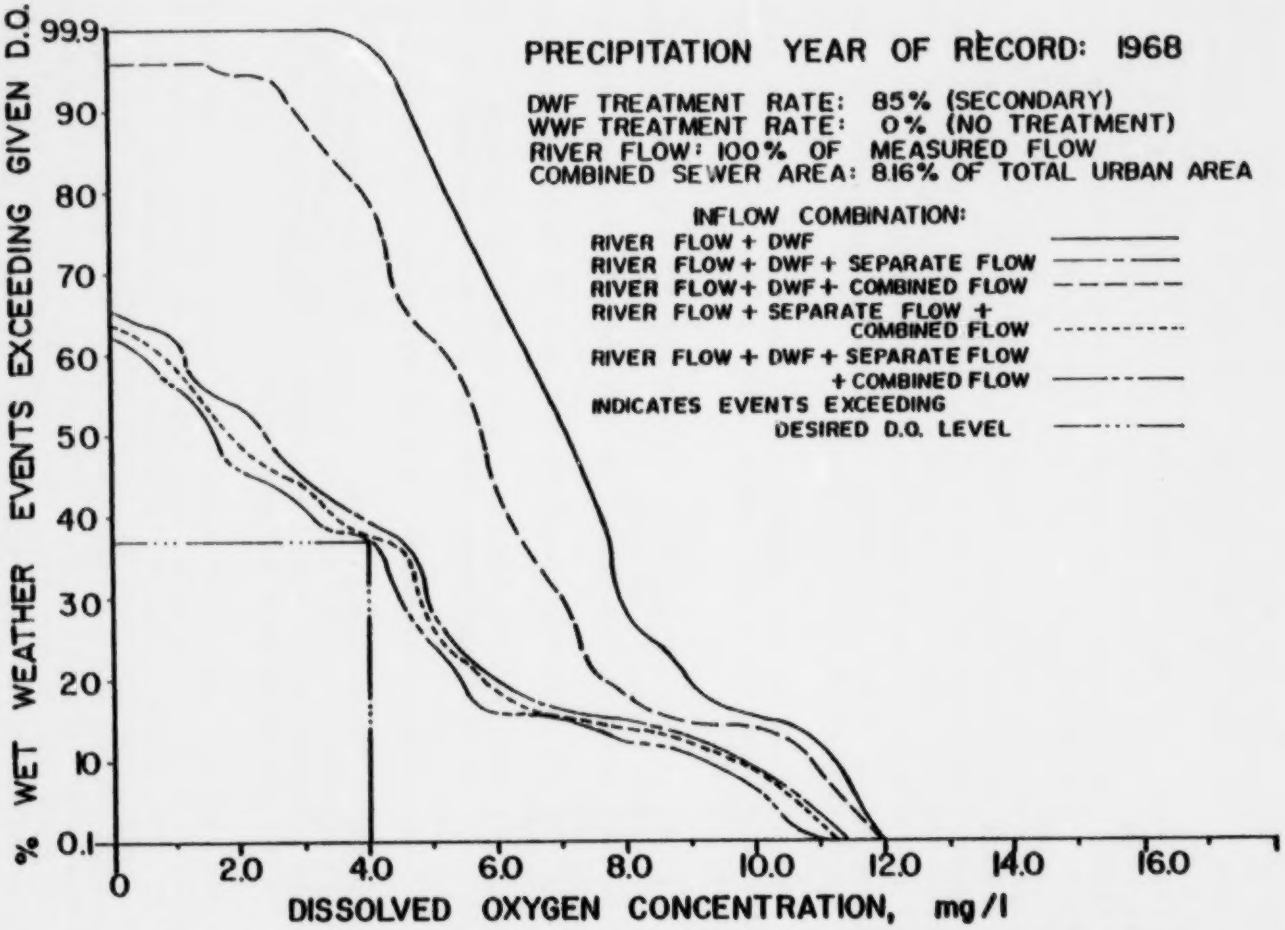


Fig. 5. Minimum DO Frequency Curves for Existing Conditions in the Des Moines River.

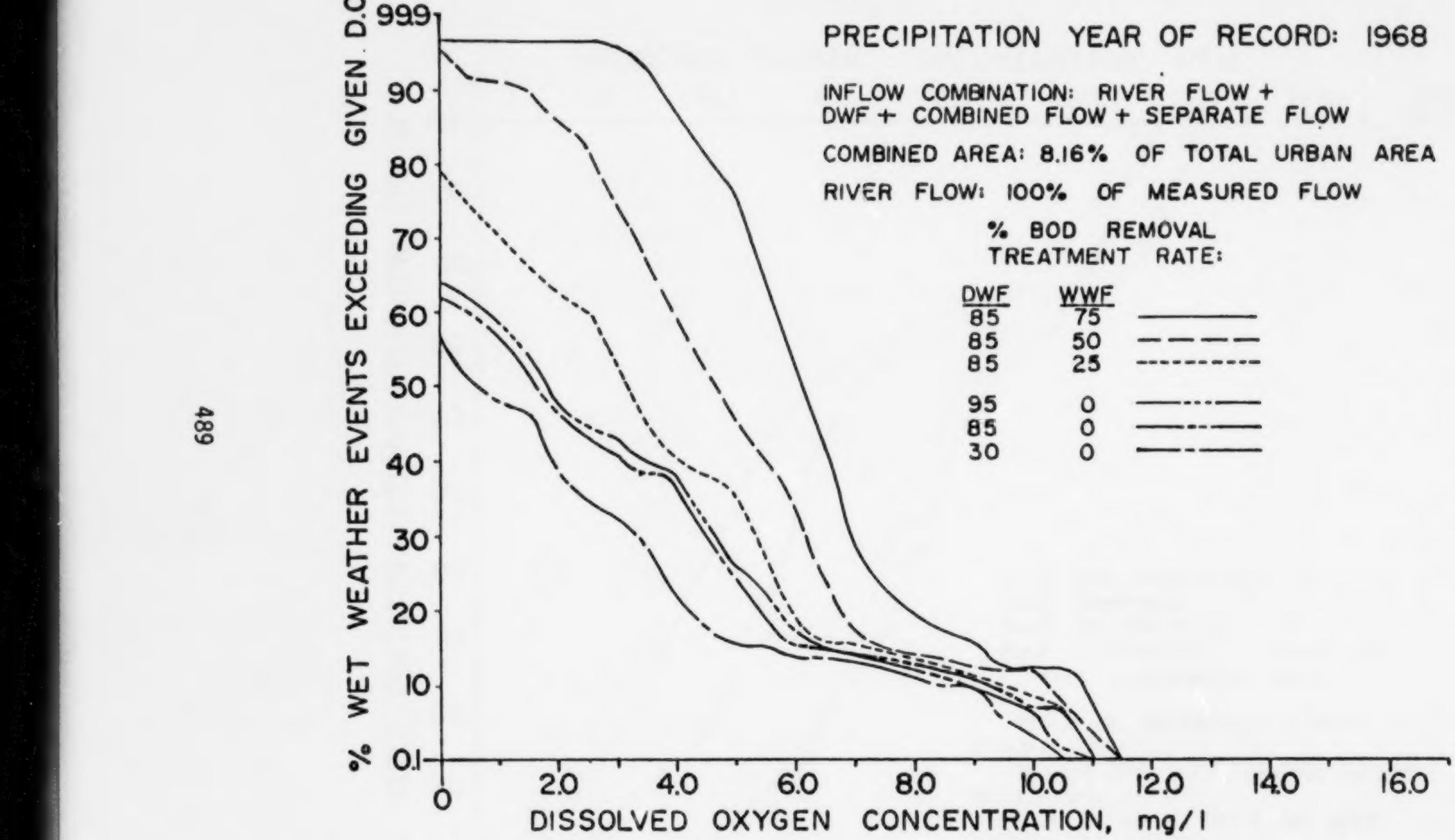


Fig. 6. Minimum DO Frequency Curves for Varied Treatment Alternatives.

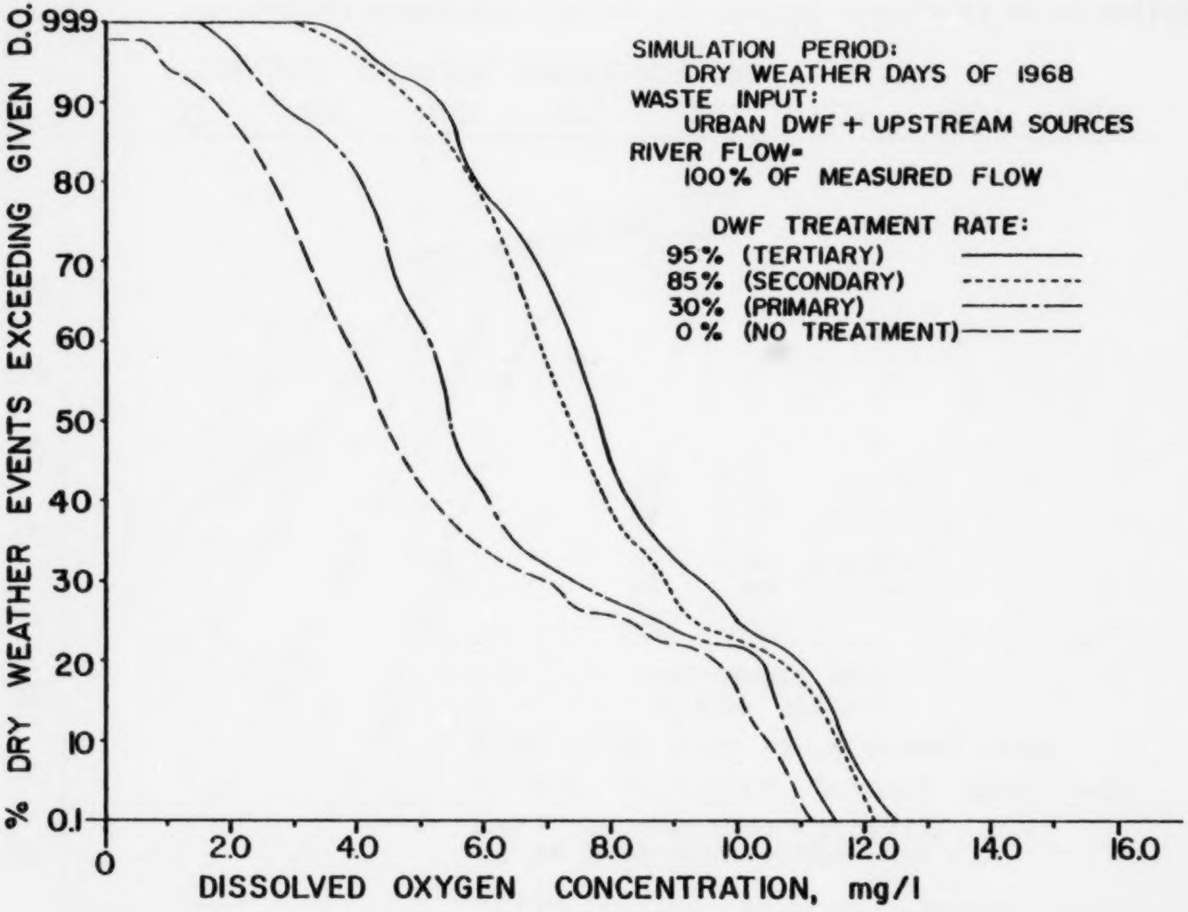


Fig. 7. Dry Weather DO Frequency Curves for Varied DWF Treatment Alternatives.

DWF secondary treatment. The Des Moines River carries a relatively high BOD load upstream of the Des Moines urban area. Thus, even during dry weather periods only, a significant increase in the DWF treatment rate does not result in a corresponding increase in the critical DO levels.

COST-EFFECTIVENESS OF CONTROL ALTERNATIVES

Six alternatives to existing DWF treatment in Des Moines during 1968 are considered in the evaluation:

1. upgrading the treatment plant from a high-rate trickling filter configuration to an activated sludge-coagulation-filtration system (85 percent BOD removal to 95 percent),
2. downgrading DWF treatment to primary (85 percent BOD removal to 30 percent), or
3. providing urban runoff control at four levels of BOD removal while maintaining secondary treatment of DWF

   (i) 25 percent,
   (ii) 50 percent,
   (iii) 75 percent, or
   (iv) 85 percent WWF treatment.

To achieve a tertiary treatment configuration, the Des Moines DWF plant must undergo an intermediate-stage modification to activated sludge. A detailed cost analysis is presented in the nationwide assessment (7). Cost estimates of providing various storage/treatment mixes for stormwater control are based on the results of a constrained optimization model (8). Total annual cost for the optimal solution can be approximated by functions of the form:

$$Z^* = ke^{\beta R_1} \quad (19)$$

where

Z^* = total annual cost, \$ per acre
k, β = parameters
R_1 = percent pollutant removal,

$$\underline{R}_1 \leq R_1 \leq \bar{R}_1$$

$\underline{R}_1$ = minimum percentage pollutant control such that the function is convex (e.g., 10 percent), and

$\bar{R}_1$ = maximum percentage pollutant removal.

For example, the cost of wet-weather control for the storm sewered area of Des Moines may be estimated by

$$Z_{ST} = 3.61\ e^{0.037\ R_1} \quad \text{. . . . .} \quad (20)$$

where

Z_{ST} = annual cost, \$ per acre of storm sewered area
R_1 = percent BOD control,

$0 \leq R_1 \leq 85.$

Similarly, the cost for the urban area served by combined sewers may be approximated by

$$Z_{CO} = 25.07\ e^{0.056\ R_1} \quad \text{. . . . .} \quad (21)$$

where

Z_{CO} = annual cost, \$ per area of combined sewered area.

A basic assumption is that a secondary treatment technology is applied to control urban runoff pollution (e.g., biological treatment, physical-chemical treatment). The total annual costs for both dry-weather and wet-weather flow control are summarized in Table 2.

The Des Moines River stretches for 200 miles (322 km) from the City of Des Moines to its junction with the Mississippi River. The entire reach is classified as a warm water "B" stream by the State of Iowa, such that the absolute minimum DO level specified is 4.0 mg/l. Figures 8 and 9 summarize the cost-effectiveness of each option during periods of urban runoff and throughout the entire simulation, respectively. Costs of wet-weather control include the annual cost of secondary treatment of DWF.

Fig. 8 illustrates that, even during periods of urban runoff, wet-weather control should be considered only after a secondary treatment technology of DWF has been achieved. However, beyond that level 25 percent and 50 percent WWF treatment become competitive with DWF tertiary treatment technology. It is clearly shown in Fig. 9 that, when the entire period of simulation (1 year) is considered, that 85 percent WWF treatment becomes asymptotic to the minimum number of violations that can be achieved, at a very high cost. Both 25 percent and 50 percent WWF treatment remain strong alternatives to higher than secondary treatment of DWF in relative cost-effectiveness.

TABLE 2. ANNUAL COSTS FOR CONTROL ALTERNATIVES

Alternative	Total Annual Cost† ($ /year)
1. Dry Weather*	
a. Activated Sludge-Coagulation-Filtration	3,507,000
b. High-Rate Trickling Filter	1,843,000
c. Primary	405,000
2. Wet Weather Control‡ (percent BOD removal)	
a. 25	816,000
b. 50	2,682,000
c. 75	9,293,000
d. 85	15,479,000

* For plant size 35.3 mgd (1.55 cu m/sec).

† Includes amortized capital costs (20 yrs, 8%, Engineering News Record index 2200), operating and maintenance costs.

‡ Based on 45,000 acres (18,211 ha) of storm sewer area; 4,000 acres (1619 ha) of combined sewer area.

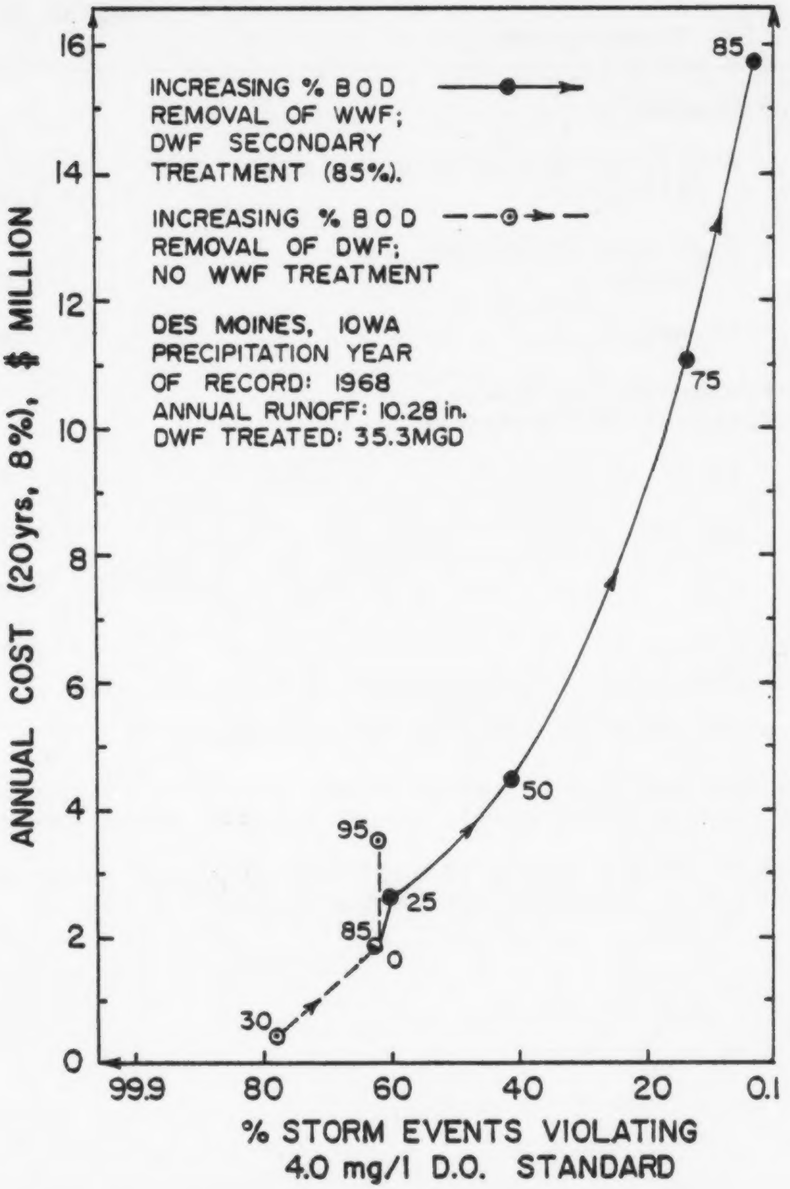


Fig. 8. Cost Effectiveness of Control Alternatives During Periods of Urban Runoff.

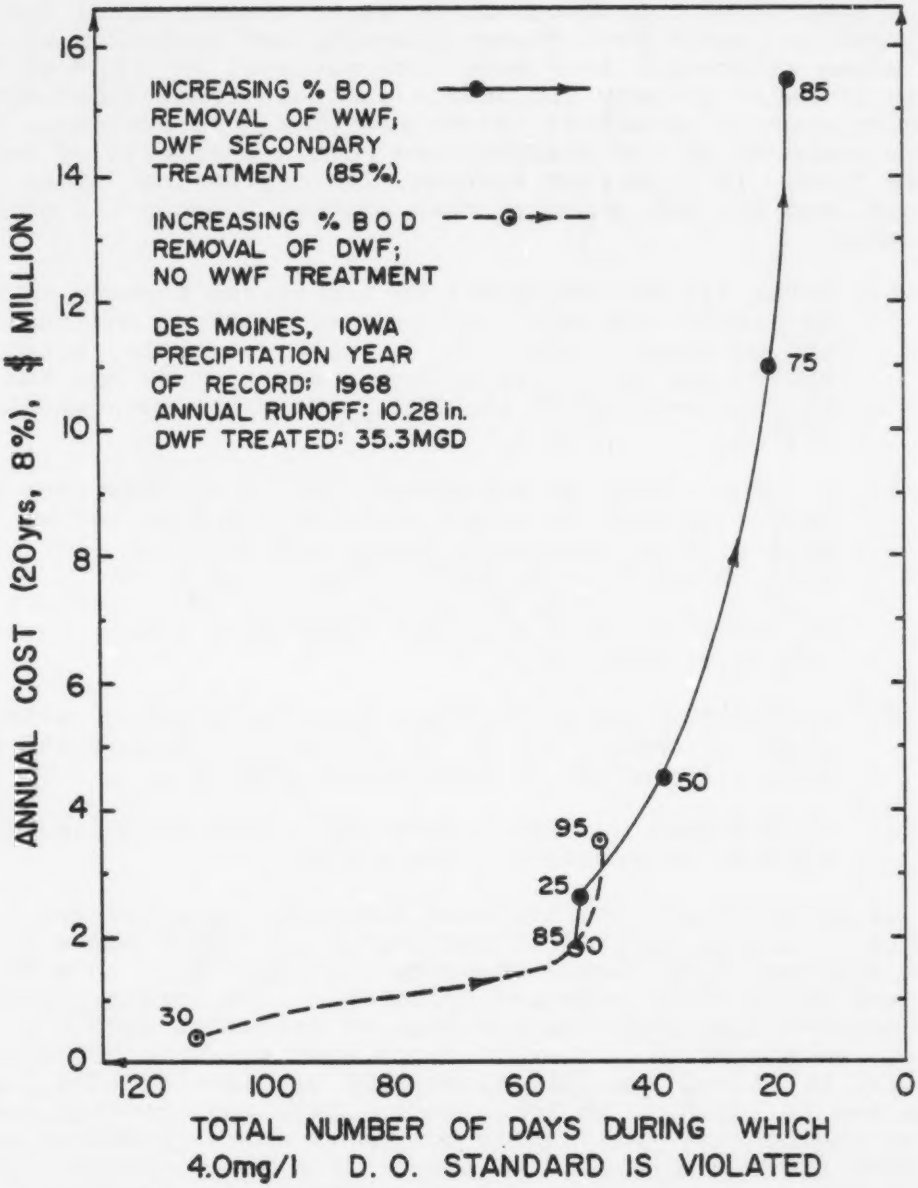


Fig. 9. Cost Effectiveness of Control Alternatives During Entire Simulation.

SUMMARY AND CONCLUSIONS

A simplified continuous river water quality model has been developed to permit preliminary planning and screening of areawide urban wastewater treatment alternatives, in terms of frequency of water quality violations and more traditional approaches such as dissolved oxygen profiles. It evaluates the stream response to the separate and combined effects of waste inputs from: 1) upstream sources, 2) dry-weather urban sources, and 3) wet-weather urban sources. Among its capabilities are:

1. Level III-Receiving may be interfaced through peripheral storage devices with various hourly, continuous urban catchment hydrologic simulation models, notably the Hydrologic Engineering Center model STORM and the continuous version of the EPA Storm Water Management Model (SWMM).
2. A large number of wastewater inflow combinations to the receiving body of water, dry-weather flow and wet-weather flow treatment rates, and upstream flow conditions may be simulated. Thus, a comparative evaluation of many urban pollution control alternatives is possible in terms of their subsequent impact on receiving water quality.
3. Continuous analysis allows representation of water quality impacts due to the random occurrence and probabilistic nature of hydrologic phenomena.
4. It computes a minimum interevent time to define statistically independent storm events.

Level III-Receiving has been developed on a general basis so that it may be applied to the surface drainage phase of most urban catchments by simply changing the input data to reflect the particular study area and hydrologic time series. There is virtually no limitation to the size of catchment modeled. In theory, an unlimited number of storm events may be processed; however, practical considerations such as computer time and costs may be limiting to some users. Data requirements are common to engineering analysis of non-point source problems and complete instructions on data preparation are provided. Field measurements, quantitative and qualitative, are nesessary to adequately calibrate model parameters and verify predicted values. The methodology is not applicable to stream and tidal river systems of such geometry and hydrodynamic behavior as to require multi-dimensional transient analysis.

An application to Des Moines, Iowa, has been presented.

Results indicate that wet-weather control should be considered after a secondary treatment technology has been achieved to control dry-weather wastewater inputs. The results are, of course, site-specific.

ACKNOWLEDGMENTS

The development of the model was supported by funds under Grant No. R-802411, Municipal Environmental Research Laboratory, Office of Research and Development, U. S. Environmental Protection Agency, Cincinnati, Ohio. Project officers were Richard Field and Chi-Yuan Fan, Storm and Combined Sewer Section, Edison, N. J. Wayne C. Huber and James P. Heaney of the University of Florida, provided invaluable suggestions during model development and also final review.

APPENDIX I.--REFERENCES

1. Anderson, R. L., "Distribution of the Serial Correlation Coefficients," Annals of Mathematical Statistics, Vol. 13, 1942, pp. 1-13.
2. ASCE, Committee on Sanitary Engineering Research, "Solubility of Atmospheric Oxygen in Water," Journal of Sanitary Engineering Division, Proc. ASCE, Vol. 86, No. SA4, July, 1960, pp. 41-53.
3. Colston, N. V., "Characterization and Treatment of Urban Land Runoff," Technical Report EPA-670/2-74-096, U. S. Environmental Protection Agency, Washington, D. C., Dec., 1974.
4. Davis, P. L., and Borchardt, F., "Combined Sewer Overflow Abatement Plan," Technical Report EPA-R-73-170, U. S. Environmental Protection Agency, Washington, D. C., April, 1974.
5. Field, R., Tafuri, A. N., and Masters, H. E., "Urban Runoff Pollution Control Technology Overview," Technical Report EPA-600/2-77-047, U. S. Environmental Protection Agency, Washington, D. C., Mar., 1977.
6. Fiering, M. B., and Jackson, B. B., "Synthetic Streamflows," Water Resources Monograph 1, American Geophysical Union, Washington, D. C., 1971.
7. Heaney, J. P., et al, "Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges, Volume II: Cost Assessment and Impacts," Technical Report EPA-600/2-77-0646, U. S. Environmental Protection Agency, Washington, D. C., Mar., 1977.
8. Heaney, J. P., Nix, S. J., and Murphy, M. P., "Storage-Treatment Mixes for Stormwater Control," Journal of the Environmental Engineering Division, ASCE, Vol. 104, No. EE4, Aug., 1978, pp. 581-592.

9. Huber, W. C., et al, "Interim Documentation, November 1977 Release of EPA SWMM," University of Florida, Gainesville, Florida, 1977.
10. Hydroscience, Inc., "Simplified Mathematical Modeling of Water Quality," U. S. Environmental Protection Agency, Mar., 1971.
11. Langbein, W. B., and Durum, W. H., "The Aeration Capacity of Streams," U. S. Geological Survey Circular 542, Washington, D. C., 1967.
12. Leopold, L. B., and Maddock, T., "The Hydraulic Geometry of Stream Channels and Some Physiographic Implications," U. S. Geological Survey Professional Paper 252, Washington, D. C., 1953.
13. Liu, Henry, "Predicting Dispersion Coefficient of Streams," Journal of the Environmental Engineering Division, ASCE, Vol. 103, No. EE1, Feb., 1977, pp. 59-69.
14. McQuivey, R. S., and Keefer, T. N., "Simple Method for Predicting Dispersion in Streams," Journal of the Environmental Engineering Division, ASCE, Vol. 100, No. EE4, Aug., 1974, pp. 997-1011.
15. Medina, M. A., "Level III: Receiving Water Quality Modeling for Urban Stormwater Management," Technical Report EPA-600/2-79-100, U. S. Environmental Protection Agency, Washington, D. C., Aug., 1979.
16. Quimpo, R. G., "Autocorrelation and Spectral Analyses in Hydrology," Journal of Hydraulics Division, Proc. ASCE, Vol. 94, No. HY2, Mar., 1968, pp. 363-373.
17. "Urban Stormwater Runoff: STORM," Generalized Computer Program 723--S8-L7520, Hydrologic Engineering Center, Corps of Engineers, July, 1976.
18. Yevjevich, V., "Stochastic Processes in Hydrology," Water Resources Publications, Fort Collins, Co., 1972.

APPENDIX II.--NOTATION

The following symbols are used in this paper:

α_1, α_2 = Regression coefficients;

BOD = Biochemical oxygen demand, mg/l;

BOD_c = Mixed BOD concentration in the combined sewer, mg/l;

BOD_d = BOD concentration of wet-weather flow treatment facility effluent, mg/l;

BOD_f = BOD concentration of municipal sewage, mg/l;

BOD_m = Mixed BOD concentration in receiving water, mg/l;

BOD_s = BOD concentration of urban stormwater runoff, mg/l;

BOD_u = Mixed BOD concentration from sources upstream of urban area, mg/l;

BOD_w = BOD concentration of treated wet-weather effluent, mg/l;

BOD_5 = Standard BOD test, 5 days at 68°F (20°C), mg/l;

β, β_1, β_2 = Regression coefficients;

C = Concentration of water quality parameter, M/L^3;

C = Concentration of dissolved oxygen (DO) in the stream, mg/l;

C_{min} = Concentration of DO at maximum deficit, mg/l;

C_s = Saturation concentration of DO, mg/l;

CBOD = Carbonaceous biochemical oxygen demand;

D = Dissolved oxygen deficit = $C_s - C$, mg/l;

D_c = Critical (maximum) deficit, mg/l;

D_o = Initial DO deficit, mg/l;

D_u = DO deficit in receiving waters upstream of inflow point, mg/l;

DO = Dissolved oxygen;

DWF = Dry weather flow, cfs;

DWFCMB = DWF contribution from combined sewer area, cfs;

DWFSEP = Dry-weather flow contribution from separate sewer area, cfs;

DWH = Number of dry-weather hours preceding each runoff event;

E = Longitudinal dispersion coefficient, feet^2 per second;

f = Self-purification ratio, K_2/K_1;

γ_1, γ_2 = Regression coefficients;

H = Stream depth, feet;

k = Number of hourly lags;

k = Coefficient;

K_1 = Deoxygenation constant of carbonaceous BOD, hours^{-1};

K_2 = Atmospheric reaeration coefficient, hours^{-1};

L_o = Remaining carbonaceous BOD concentration, mg/l;

n = Total number of data points or observations of a hydrologic process;

Q = Streamflow, cfs;

Q_c = Combined sewer flow, cfs;

Q_d = DWF treated effluent, cfs;

Q_s = Urban runoff carried by the separate storm sewer, cfs;

Q_w = Wet-weather flow (WWF) treated effluent, cfs;

R_d = Fraction removal of BOD achieved by the DWF treatment facility;

$r_I(k)$ = Sample estimate of lag-k autocorrelation coefficient for rainfall;

R_o = Deficit load ratio = D_o/L_o;

$r_Q(k)$ = Sample estimate of lag-k autocorrelation coefficient for runoff;

R_w = Fraction removal of BOD achieved by the WWF treatment facility;

R_1 = percent pollutant removal;

$\underline{R}_1$ = minimum percent pollutant removal;

$\bar{R}_1$ = maximum percent pollutant removal;

S = Sources and sinks of the substance C, M/L^3T;

t_α = Standardized normal variate;

t_c = Elapsed time at which critical deficit occurs, hours or days;

T = Stream temperature, °C;

$TL\ [r_I(k)]$ = Tolerance limits at a 95 percent probability level;

U = Flow velocity in stream, feet per second;

WWF = Wet-weather flow, cfs;

x = Distance downstream, feet or miles;

x_i = Discrete data series (observations) of a hydrologic process;

X = A hydrologic event;

Z^* = total annual cost, $ per acre;

Z_{ST} = annual cost, $ per acre of storm sewered area;

Z_{CO} = annual cost, $ per acre of combined sewered area.

POTENTIAL OF URBAN STORMWATER IMPACTS BASED ON COMPARATIVE ANALYSIS OF WET AND DRY WEATHER POLLUTANT LOADS.

by

Douglas Ammon* and Richard Field**

INTRODUCTION

Urban stormwater discharges and combined sewer overflows are significant contributors of many pollutants to receiving waters (1,2,3,4). Whether or not these pollutant loads cause severe enough adverse impacts to justify some degree of control must ultimately be decided with site specific evaluations. However, characterization of urban stormwater discharges and combined sewer overflows, in terms of concentrations and loads, provides useful rough-cut, order of magnitude indications of potential receiving water impacts. Thus, summaries of urban wet-weather characterization data are presented for several groups of parameters including heavy metals, petroleum hydrocarbons, biochemical oxygen demand (BOD) and chemical oxygen demand (COD). Heavy metals and petroleum hydrocarbons are selected because of their abundance in urban wet-weather discharges and the growing concern with toxic substances in the environment. The oxygen demand parameters are included because dissolved oxygen levels in surface waters remain the most important and popular indicator of water quality. Furthermore, we should not become too preoccupied with toxicity issues that we lose sight of other important concerns, such as dissolved oxygen depletion. There are many constituent groups that are not included in this paper (e.g., chlorinated hydrocarbons, nutrients, solids, pathogenic indicators, etc.). The exclusion of these constituent groups is not meant to diminish their importance.

Comparisons are presented between the various pollutant loads from urban wet-weather discharges and from publicly owned treatment works (POTW) effluents to indicate the relative importance of each source. The significance of each pollutant is identified based on loading potentials or, whenever possible, on exceedance of water quality criteria for aquatic life protection.

---

*Staff Engineer, Storm & Combined Sewer Section, Municipal Environmental Research Laboratory (Cincinnati), U.S. Environmental Protection Agency, Edison, New Jersey 08817

**Chief, Storm & Combined Sewer Section, Municipal Environmental Research Laboratory (Cincinnati), U.S. Environmental Protection Agency, Edison, New Jersey 08817

OXYGEN CONSUMING MATERIAL

Urban stormwater discharges and combined sewer overflows contribute a large portion of the oxygen consuming material to receiving waters (1,2,3,4). The average dry-weather flows (sanitary flows) for the urbanized area in the United States have been estimated to be about 0.61 Mgal/yr/acre (5700 m^3/yr/ha) and 0.48 Mgal/yr/acre (4500 m^3/yr/ha) for combined and separate sewered areas, respectively (5). The average wet-weather flows for the urbanized area in the United States have been estimated to be about 0.45 Mgal/yr/acre (4200 m^3/yr/ha) and 0.40 Mgal/yr/acre (3800 m^3/yr/ha) for combined sewered and storm sewered areas, respectively (5). The annual wet-weather flows vary between 0.08 - 0.95 Mgal/yr/acre (750 - 8900 m^3/yr/ha) for combined sewered areas and between 0.05 - 0.68 Mgal/yr/acre (470 - 6400 m^3/yr/ha) for separate sewered areas depending on the characteristics of the particular urbanized area. The BOD concentrations in combined sewer overflows are approximately one-half the raw sanitary sewage BOD or about 100 mg/l (3); however, concentrations have been reported as high as 640 mg/l (3). The BOD concentrations in urban stormwater discharges are about equal to those of secondary effluents (3); however, concentrations as high as 885 mg/l have been reported (2). The reader may wish to consult references by Huber et al. (2), Lager et al. (3), and Manning et al. (4) for additional characterization information for urban wet-weather discharges.

Using the above approximate concentrations and flows, combined sewer overflows are calculated to contribute about 380 lb/acre/yr (425 kg/ha/yr) of BOD which is about 28 percent of the annual BOD load from the combined sewered area given no treatment of the dry-weather flow. If primary and secondary treatment are provided for the dry-weather flow, then the combined sewer overflows will contribute 35 percent and 72 percent of the annual BOD load to the receiving water, respectively. Likewise, in separate sewered areas, urban stormwater discharges contribute about 70 lb/acre/yr (75 kg/ha/yr) of BOD which is about 45 percent of the annual BOD load if secondary treatment is provided for the dry-weather flow. Table 1 shows the national annual urban wet- and dry-weather BOD and COD loads for the developed urban area of the United States assuming secondary treatment of the dry-weather flows.

These values represent the annual BOD loads to receiving waters based on approximate "national" averages. Short-term effects of oxygen consuming material resulting from wet-weather discharges are the more important concerns. Obviously, annual loads are distributed intermittently to the receiving water by storm events. Thus, the relative magnitudes of urban wet-weather loads compared to the POTW discharges are much greater during and following storm events. Furthermore, concentrations of oxygen consuming materials are usually unevenly distributed within a storm event with greater concentrations often occurring in the first portions of the storm, the so-called first-flush phenomenon. Depending on the receiving water, reaeration, dilution, etc., this may result in shock effects that may be detrimental to aquatic populations. A recent analysis of 83 water quality monitoring sites in and downstream of urban areas found that 42 percent demonstrated a 60 percent or greater probability of higher than average DO deficit occurring at times of higher than average streamflow or on days with rainfall (6). This was found despite the fact that the monitoring sites were not located specifically to observe urban wet-weather DO deficits. Roughly 25 percent of the 83 monitoring sites would not meet a 2.0 mg/l standard, although the frequency of violations would generally be less than 6 times per year.

Urban stormwater discharges contain oxygen consuming materials which are not as readily biodegradable as combined sewer overflows and POTW effluents. COD:BOD ratios for urban stormwater discharges and combined sewer overflows are presented in Figure 1 (COD is taken as a crude surrogate of ultimate oxygen demand). Secondary effluents typically have ratios around 2. Higher ratios indicate that oxygen consumption may occur long after the storm events. Thus, "background" oxygen demand observed during dry-weather periods are partially due to wet-weather discharges. Another related characteristic of urban wet-weather discharges is the high solids fractions, and associated oxygen demand, relative to the POTW effluents. For instance, combined sewer overflows, in addition to the solids from the stormwater portion, have a raw sanitary portion and flushed "dry-weather" deposition which are high in settleable solids and oxygen demand. Much of the solids from the urban wet-weather discharges with their associated pollutants will settle in the receiving water at outfalls and low-velocity regimes contributing to such problems as high sediment oxygen consumption, physical blanketing (loss of habitats) and benthic toxicity.

HEAVY METALS

Urban runoff is generally recognized as a major means of transporting heavy metals to receiving waters. A major source of most heavy metals in urban runoff is transportation activity. Other sources of heavy metals include atmospheric particulate fallout and washout (from industrial stacks, incinerators, open fires and metal smelters), metal plating, soil erosion, vegetative material, pesticides, chemical spills, and natural background. Municipal and industrial wastewaters are additional sources of heavy metals in combined sewer overflows and in storm sewer discharges with extraneous connections.

Heavy metals may be classified as those essential to life and those not known to be essential to life processes. The hypothetical relationships among population effects (lethal and sublethal), heavy metal concentration and duration of exposure show increasingly beneficial effects (biostimulation) up to a certain optimum region (i.e., combinations of concentrations and durations). Beyond this region, there are tolerance regions which may be either relatively narrow or broad, beyond which benefits decrease, injurious effects begin, and finally lethal doses are reached. For those heavy metals not known to be essential, the first region is missing, but there is generally a tolerance region followed by the toxic and the lethal dose regions. Of greatest concern are those metals known to be toxic at low or moderate concentrations and at relatively short duration of exposure to a variety of aquatic species or to man. Currently, there are 13 heavy metals on the EPA priority pollutant list, some of which may fall into this last category for urban wet-weather discharges. The 13 heavy metals include: antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, and zinc.

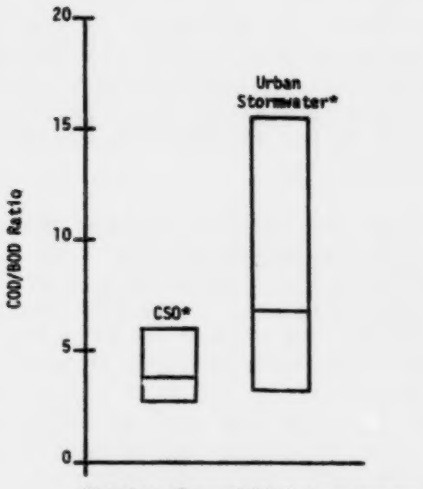
TABLE 1. NATIONAL ANNUAL URBAN WET- AND DRY-WEATHER BOD AND COD LOAD COMPARISONS.*

Type	Percent of Developed Area	Annual DWF**		Annual WWF**		Percent WWF	
		BOD (mil. lb.)	COD (mil. lb.)	BOD (mil. lb.)	COD (mil. lb.)	BOD	COD
Combined Sewered	14.3	340	910	880	2640	72	74
Storm Sewered	38.3	710	1890	600	2500	45	57
Unsewered	47.4	310	830	360	1800	54	68
Total	100	1330	3630	1840	6940	58	67

**lb. = 0.454 kg

Assuming*:

	BOD (mg/l)	COD (mg/l)
CSO	100	300
Stormwater (Sewered)	30	125
Stormwater (Unsewered)	20	100
Dry-Weather	30	80



*Based on flow-weighted storm average concentrations from 7 CSO locations and 15 urban stormwater locations.

Center line is the overall average

FIGURE 1. RATIOS OF COD TO BOD FOR URBAN WET-WEATHER DISCHARGES.

Heavy metals in urban runoff have been investigated at numerous sites across the country for more than a decade. Some of the results of these sampling programs are summarized in Table 2 for combined sewer overflows and urban stormwater discharges. The overall ranges of heavy metal concentrations from the various locations are presented in Figures 2 and 3 along with the overall range of POTW effluent concentrations from a national survey (7). As illustrated, the heavy metal concentrations are highly variable.

For discussion purposes and for comparison with other sources, "typical" average metal concentrations are needed, although any use of typical values must be tempered with the usual caveats (e.g., variability of urban runoff quality). The assumed typical average metal concentrations for combined sewer overflows and urban stormwater discharges are presented in Table 3. The typical average metal concentrations are based on the arithmetic average of flow-weighted storm average concentrations from locations that report them in that manner. For example, the typical average lead concentration for urban stormwater is 160 μg/l which is based on flow-weighted storm averages from 9 locations and a total of 126 storm events. For some of the metals there are too few locations with flow-weighted data to make statistically reasonable estimates. In these cases the typical average metal concentrations are approximated by the logorithmic mid-point between the overall minimum and maximum concentrations.

EPA has recently published draft water quality criteria for most of the pollutants originally listed as toxic under the Clean Water Act (Section 304, PL 95-217) including 6 of the 13 metals on the priority pollutant list (18, 19). These water quality criteria are estimates of the concentrations of water constituents in ambient water which, when not exceeded, will ensure a water quality sufficient to protect a specified water use. Under the Act, these water quality criteria are scientific entities based solely on data and scientific judgment. They do not necessarily reflect considerations of economic or technological feasibility.

Figure 4 shows examples of the 24-hour average and ceiling water quality criteria for the protection of freshwater aquatic life for lead. These water quality criteria and the criteria for 4 other metals are functions of the hardness in the ambient receiving water. Using the typical average metal concentrations presented earlier, the number of dilutions of urban stormwater discharges and combined sewer overflows required to meet the 24-hour average water quality criteria are presented in Tables 4 and 5, respectively. Arbitrary hardness values of 50, 100, and 200 mg/l as $CaCO_3$ are assumed, which indicates the effects over a reasonable range of receiving water hardness. Clearly, these 6 metals are in sufficient concentrations to potentially affect aquatic life, particularly cadmium and copper. Water quality criteria have not been established for the other heavy metals on the priority pollutant list; however, these metals are usually present in urban runoff in concentrations at least greater than detection limits.

This analysis assumes that the metals will remain in the water column and are in available forms. However, many of the metals are associated in

TABLE 2. SUMMARY OF HEAVY METAL CONCENTRATIONS FOR COMBINED SEWER OVERFLOWS AND URBAN STORMWATER DISCHARGES.

Arsenic (mg/l)	Cadmium (µg/l)	Chromium (mg/l)	Copper (mg/l)	Lead (mg/l)	Mercury (µg/l)	Nickel (mg/l)	Zinc (mg/l)	Type*	Location	Ref.
---	≤10.	≤0.05	≤0.54	≤0.20	---	≤0.01	0.04-0.53	CSO (a)	Syracuse, NY	2
---	<10. (3)	≤0.03 (4)		<0.02 (3)	1.5-28. (2)	<0.02 (3)	0.07-0.11 (2)	CSO (a)	Rochester, NY	9
---	10.-180. m=30 (3)	0.1-18. m=2.2 (3)	0.09-0.55 m=0.16 (5)	0.1-33.0 m=7.5 (3)	---	---	0.1-1.4 m=0.4 (5)	CSO (a)	Lancaster, PA	2
0.049-0.060 m=0.050 (3)	3.9-1600. m=27 (5)	0.01-4.2 m=0.60 (5)	0.02-4.2 m=0.35 (5)	0.09-1.8 m=0.30 (5)	---	---	0.15-5.3 m=0.93 (5)	CSO (a)	Seattle Wash.	2
3.1-14. (5)	0.7-95.0 (16)	0.1-0.9 (16)	0.1-0.68 (16)	0.2-1.28 (16)	0.1-3.1 (16)	0.08-0.50 (13)	0.12-0.80 (15)	CSO (c)	Newton Creek WTP, New York City	10
---	---	---	0.062 (3)	0.08 (3)	2.5 (3)	---	0.161 (3)	CSO (b)	Lake Washington Seattle, Wash.	11
---	---	---	---	0.0-1.6 (176)	0.05-47. (76)	---	---	US (d)	Knoxville, Tenn.	12
---	---	---	---	0.05-0.8 (59)	---	---	---	US (d)	Atlanta, GA	13
---	---	0.06-0.47 (232)	0.04-0.50 (225)	0.1-2.86 (336)	---	0.09-0.29 (103)	0.09-4.6 (310)	US (d)	Durham, NC	1
0.0028	6.	0.0025	0.27	0.2	0.1	0.024	0.465	US (d)	Gothenborg, Sweden	14
---	1.-6. m=1.3 (5)	0.01-0.02 m=0.01 (7)	≤0.04 (7)	0.03-1.1 m=0.16 (7)	---	---	0.01-0.56 m=0.077 (7)	US (a)	Broward County, FL	2
---	---		---	0.1-1.5	---	---	---	US (a)	Durham, NC	2
0.05-0.1 m=0.05 (3)	0.4-10.0 m=0.4 (26)	0.009-0.1 m=0.02 (7)	0.01-0.23 m=0.02 (7)	0.002-3.0 m=0.14 (26)	---	---	0.005-0.99 m=0.07 (26)	US (a)	View Ridge 1, Seattle, Wash.	2
0.05 (2)	3.9-11.0 m=4.1 (5)	<0.01 (5)	0.01-0.20 m=0.05 (5)	0.1-2.5 m=0.17 (5)	---	---	0.005-0.64 m=0.05 (5)	US (a)	View Ridge 2, Seattle, Wash.	2
0.04-0.1 m=0.05 (3)	3.9-40. m=4.9 (26)	0.009-0.09 m=0.02 (7)	0.004-0.4 m=0.12 (7)	0.01-1.4 m=0.2 (26)	---	---	0.017-2.3 m=0.23 (26)	US (a)	South Seattle, Seattle, Wash.	2
0.045-0.070 m=0.05 (3)	0.6-7000. m=20.8 (23)	0.-09-11.0 m=0.1 (7)	0.01-0.31 m=0.07 (7)	0.01-2.7 m=0.2 (23)	---	---	0.01-1.7 m=0.14 (22)	US (a)	South Center, Seattle, Wash.	2
0.049-0.060 m=0.05 (3)	3.9-8.0 m=4.4 (5)	0.009-0.21 m=0.01 (5)	0.009-0.35 m=0.08 (5)	0.09-2.5 m=0.21 (5)	---	---	0.01-0.49 m=0.06 (5)	US (a)	Lake Hills, Seattle, Wash.	2
0.050 m=0.05 (2)	3.9-10.0 m=4.5 (4)	0.009-0.01 m=0.01 (4)	0.009-0.35 m=0.16 (3)	0.009-0.27 m=0.10 (4)	---	---	0.009-0.18 m=0.04 (4)	US (a)	Highlands, Seattle, Wash.	2
---	---	---	---	0.009-0.80 m=0.18 (4)	---	---	0.052-0.57 m=0.20 (5)	US (a)	Greenfield, Mass.	2
---	---	---	---	0.001-0.51 m=0.09 (6)	---	---	0.01-0.39 m=0.10 (6)	US (a)	Northampton, Mass.	2
---	---	---	---	0.00-2.23 (12)	---	---	---	US (d)	Oklahoma City Oklahoma	3
---	<2.-6 (11)	0.005-0.04 (11)	0.01-0.09 (11)	0.1-1.5 (11)	<0.1-0.6 (11)	---	0.06-0.55 (11)	US (d)	San Jose, Calif.	15
0.04 (1)	58. (1)	0.07 (1)	0.13 (1)	4.94 (1)	---	0.05 (1)	0.68 (1)	US (b)	Lake Eola Orlando, Fl	16

*
CSO = Combined Sewer Overflow
US = Urban Stormwater
(a) = Flow weighted storm averages
(b) = Storm averages
(c) = Composite samples
(d) = Grab samples

The numbers in parenthesis refer to the number of samples or the number of storm averages if reported.

m = mean concentration (flow-weighted).

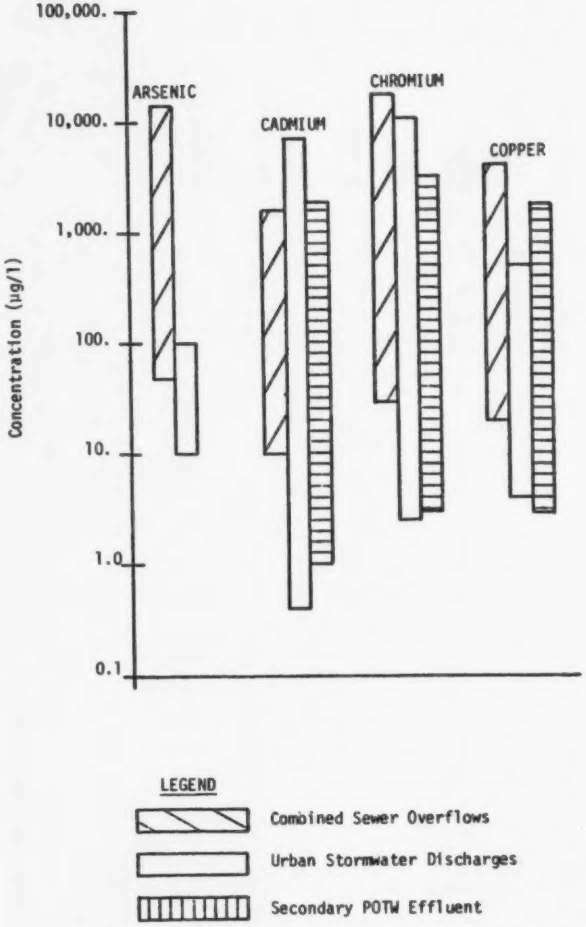


FIGURE 2. CONCENTRATION RANGES OF ARSENIC, CADMIUM, CHROMIUM AND COPPER IN URBAN DISCHARGES.

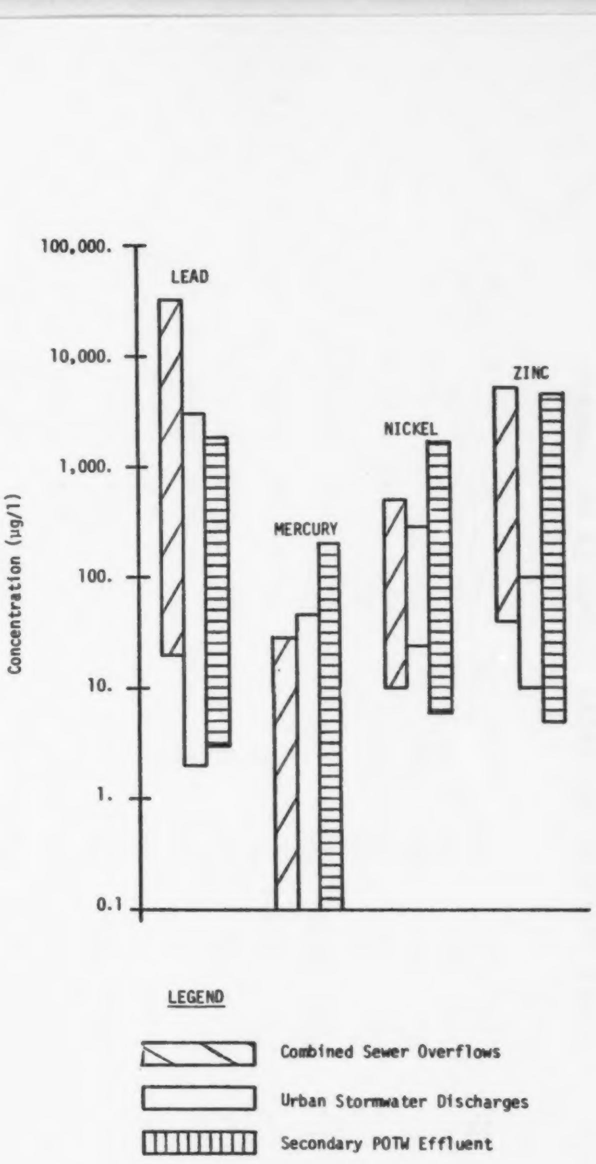


FIGURE 3. CONCENTRATION RANGES OF LEAD, MERCURY, NICKEL AND ZINC IN URBAN DISCHARGES.

TABLE 3. ASSUMED TYPICAL AVERAGE HEAVY METAL CONCENTRATIONS FOR COMBINED SEWER OVERFLOWS AND URBAN STORMWATER DISCHARGES.

	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Combined Sewer Overflows (μg/l)	830	126	730	290	810	1.7	70	460
Urban Stormwater (μg/l)	50*	5.8*	170*	40*	160*	2.2	80	150*

* Based on Average of Flow-Weighted Storm Average Concentrations From 5 or More Locations.

	Antimony	Beryllium	Selenium	Silver	Thallium
Urban Stormwater (μg/l)**	20	2	3	8	2

** 1 Grab Sample
San Jose, California (17)

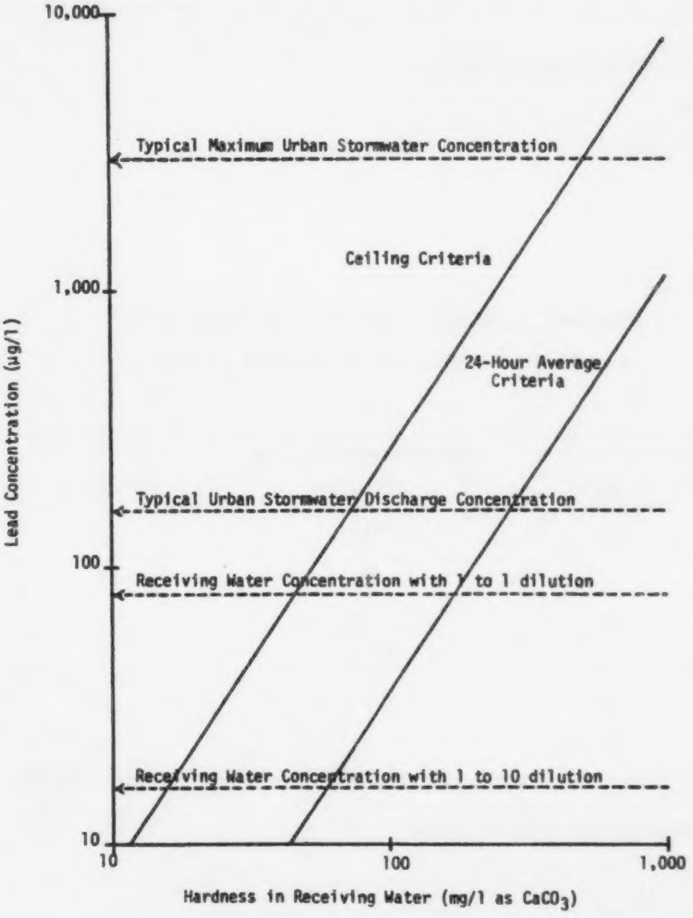


FIGURE 4. PROPOSED FRESH WATER QUALITY CRITERIA FOR PROTECTION OF AQUATIC LIFE FOR LEAD

TABLE 4. DILUTIONS OF URBAN STORMWATER DISCHARGES NEEDED TO MEET 24-HOUR AVERAGE WATER QUALITY CRITERIA (AQUATIC LIFE).

	Freshwater Aquatic Life			
Metal	50 mg/l*	100 mg/l*	200 mg/l*	Saltwater Aquatic Life
Cadmium	15	8	5	6
Copper	44	28	18	100
Lead	13	4	2	-
Nickel	4	2	1	-
Silver	890++	890++	890++	31
Zinc	4	3	2	-

* Ambient Receiving Water Hardness as $CaCO_3$

++ Only One Grab Sample

- Criteria Not Established

TABLE 5. DILUTIONS OF COMBINED SEWER OVERFLOW NEEDED TO MEET 24-HOUR AVERAGE WATER QUALITY CRITERIA (AQUATIC LIFE).

	Freshwater Aquatic Life			
Metal	50 mg/l*	100 mg/l*	200 mg/l*	Saltwater Aquatic Life
Cadmium	335	183	100	125
Copper	159	101	64	370
Lead	64	22	8	-
Nickel	4	2	1	-
Silver	+	+	+	+
Zinc	17	11	7	-

* Ambient Receiving Water Hardness as $CaCO_3$

+ Insufficient Data

- Criteria Not Established

varying degrees with particulates; therefore, bottom accumulation will occur in low velocity regions in the receiving water. Thus, the most probable impacts will occur in the sediments. These impacts are not reflected by the water quality criteria. The heavy metals associated with solids, either several sources of solids in urban wet-weather discharges or digested treatment plant sludges, are presented in Table 6. Metals associated with street surface dust and dirt and with soils are two sources of metals in urban stormwater discharges. In addition to these two sources and other sources, combined sewer overflows can contain significant quantities of solids which are deposited in the sewer during dry-weather and subsequently flushed out by the higher wet-weather flows. The metal concentrations associated with this dry-weather deposition in the Boston area are presented in Table 6 (22).

There are studies that have investigated metal accumulation in the sediments. For example, Pitt and Bozeman (24) found lead concentrations in the sediments to be about 10 times greater in the urban reaches than in the nonurban reaches of Goyote Creek, which receives minimal pollutant discharges, except for urban runoff. Lead concentrations in urban samples of algae, crayfish and cattails were 2 to 3 times greater than in nonurban samples. Zinc concentrations in urban algae and cattail samples were about 3 times greater. The nonurbanized section of the creek was found to support a comparatively diverse assemblage of aquatic organisms including at least 12 species of fish and various benthic macroinvertebrate taxa. In contrast, the portion receiving urban runoff was found to support an aquatic community that is lacking in diversity and dominated by pollution tolerant organisms. Although specific cause and effect relationships are unclear, it is probably safe to conclude that pollutional stresses from the urban wet-weather discharges are the major contributing factor.

Wilbur and Hunter (25) investigated bottom sediments in the Saddle River upstream and downstream of Lodi, New Jersey. Metal concentrations increased in the downstream sediment samples. The average metal enrichments are presented in Table 7. As expected in cross-sectional sediment samples metal concentrations were the highest in the regions of relatively low velocity and lowest in the high velocity regions.

Average effluent concentrations from secondary POTW's along with the typical average metal concentrations for urban wet-weather discharges are presented in Table 8. Concentration ratios of combined sewer overflows to secondary POTW effluents and urban stormwater discharges to secondary POTW effluents are also presented. Cadmium, chromium, lead and zinc concentrations are typically greater in combined sewer overflows than in secondary POTW effluents while cadmium, chromium and lead are greater in urban stormwater than in secondary POTW effluents. In all cases, wet-weather metal concentrations are significant.

Comparisons of heavy metal loads in combined sewer overflows and urban stormwater discharges with the secondary POTW effluent loads are presented in Table 9. The annual loads are estimated from the United States average annual wet- and dry-weather flows for combined sewered, storm sewered and unsewered

TABLE 6. HEAVY METALS ASSOCIATED WITH VARIOUS SOLIDS (mg/kg dry solids).

Type	Ref.	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc	Comment
Metals concentrations in sludges from 8 CSO treatment plants	20	---	---	52-2471 m=787 (8)	200-1454 m=636 (8)	164-2448 m=1100 (8)	0.01-100.5 m=2.2 (8)	83-995 m=372 (8)	697-7154 m=1700 (8)	Average % solids:~1%
Metals concentrations in sludges from 33 POTW's	21	---	m=30	m=1370	m=860	m=1380	m=15	m=580	m=1960	Average % solids: 2-7%
Metals concentrations in dry-weather deposition flushed from sewers	22	---	2-19 (68)	3-98 (65)	47-2670 (68)	8-680 (68)	1-14 (39)	5-85 (68)	13-2264 (68)	------------
Metals concentrations in soil.	23	0.1-40 m=6	0.01-0.7 m=0.06	5-3000 m=100	2-100 m=20	2-200 m=10	0.01-0.3 m=0.03	10-1000 m=40	10-300 m=50	------------
Metals concentrations in street surface Dust and Dirt	4	0 (3)	0-11 m=3 (57)	10-480 m=180 (65)	25-810 m=90 (65)	0-7600 m=1970 (64)	0.0-0.1 m=0.02 (6)	1-170 m=62 (65)	90-3040 m=470 (65)	Typical Dust and Dirt Accumulation: 45kg/curb-km/day

The numbers in parentheses refer to the number of samples.

m = mean concentration (mg/kg).

TABLE 7 METAL ENRICHMENT IN SEDIMENTS IN THE SADDLE RIVER (25).

	Cadmium	Chromium	Copper	Lead	Nickel	Zinc
Average Metal Enrichment	5.2	5.1	3.1	6.7	2.8	3.5
Highest Sediment Concentration Upstream of Urban Area (mg/kg)	0.4	6.5	28.9	38.6	6.5	84.2
Highest Sediment Concentration Downstream of Urban Area (mg/kg)	2.9	34.9	104.8	200.	22.3	275.1

TABLE 8. AVERAGE HEAVY METALS IN COMBINED SEWER OVERFLOWS, URBAN STORMWATER DISCHARGES, AND SECONDARY POTW EFFLUENT.

	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Typical Average CSO (μg/l)	126	730	290	810	1.7	70	460
Typical Average Urban Stormwater (μg/l)	53	170	80	160	2.2	80	110
Average Secondary POTW Effluents (μg/l) (7)	32	220	570	90	4.1	180	280
Ratio CSO to POTW Concentrations	3.9	3.3	0.5	9.0	0.4	0.4	1.6
Ratio Urban Stormwater to POTW Concentrations	1.6	0.8	0.1	1.8	0.5	0.4	0.4

TABLE 9. RATIOS OF HEAVY METAL MASS LOADS IN WET-WEATHER DISCHARGES TO SECONDARY POTW EFFLUENTS.

	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Annual CSO/POTWs*	2.9	2.4	0.4	6.6	0.3	0.3	1.2
Annual Urban Stormwater/POTWs**	1.3	0.7	0.1	1.5	0.4	0.3	0.3
Storm Event CSO/POTWs*	39	33	5	90	4	4	16
Storm Event Urban Stormwater/POTWs**	16	8	1	18	5	4	4

*For Combined Sewered Areas (14.3% of the U.S. developed urban area)

**For Separate Sewered Areas (38.3% of the U.S. developed urban area) areas, as determined by Heaney et al. (5). These ratios will probably continue to increase (that is, the relative contribution of urban wet-weather sources will increase) in the future as POTW's are upgraded and as the federal pretreatment program reduces industrial contributions to POTW's. However, lead may be the exception since leaded gas usage should decline. The storm event load comparisons are based on wet-weather flows assumed to be ten times greater than the dry-weather flows which is not an unlikely storm event. For combined sewered areas annual loads of cadmium, chromium, lead and zinc are greater from combined sewer overflows than in the secondary POTW effluents. For storm sewered areas annual loads of cadmium and lead are greater than the secondary POTW effluents.

PETROLEUM HYDROCARBONS

Petroleum enters urban runoff from many dispersed sources; however, it may be attributed primarily to transportation activity (spills, leaks and disposal of motor vehicle lubricants, antifreeze and hydraulic fluids). For instance, about one-half billion gallons (2 million m^3) of automotive lubricants are lost to the environment annually (1972) in the United States (26).

Petroleum hydrocarbons, particularly the polynuclear aromatic hydrocarbons (PAH's), have been shown to be carcinogenic and mutagenic in mammalian and microbial systems. There is also evidence that PAH's produce cancerous growths in some aquatic invertebrates and vertebrates, especially early life forms (e.g., spot shrimp larvae). The various PAH's that are on the EPA priority pollutant list are presented in Table 10.

Direct urban stormwater discharges and municipal waste disposal (wastewater effluents and sludge dumping) have been estimated to contribute about 10 percent of the petroleum hydrocarbons entering the ocean on a global basis as shown in Table 11 (27). This may be put into further perspective when considering that both these sources contribute the major chronic portion to adjacent coastal and estuarine areas which, on a mass balance basis, contribute more petroleum hydrocarbons to these near-shore environments than oil spills. It follows that urban runoff contributes a major portion of the petroleum hydrocarbon loads to fresh waters. For instance, urban runoff has been reported to be the major contributor of petroleum hydrocarbons to Lake Washington (28).

The presence of specific petroleum derived hydrocarbons in urban runoff has only been investigated by a few (28, 29, 30). What has been commonly reported in urban runoff sampling programs are oil and grease, usually by hexane extractable materials (HEM's). For instance, in Jamaica Bay, New York, about 50 percent of the HEM's are contributed by combined sewer overflows (31). Oil and grease cover a broad spectrum of organic material, including fatty acids, soaps, esters, fats, waxes, and various petroleum hydrocarbons. Thus, they should only be considered gross indicators of the presence of petroleum hydrocarbons since they do not distinguish between origins (i.e., biogenic versus petroleum origins). However, street solids (dust and dirt fraction) in the Washington, D.C. area have been reported to contain 6400 mg/kg of oil and grease of which 3600 mg/kg are petroleum hydrocarbons (32). Dust and dirt is a major solids component in urban runoff.

A study in Lake Washington investigated contaminant concentrations in sediments (11). Table 12 shows the linear correlation matrix of oil and grease with several other parameters in sediments near combined sewer and storm sewer outfalls and in control areas. Interestingly, the oil and grease at the combined sewer outfalls correlate well with TOC while the correlations are much weaker at the storm sewer outfalls. Likewise, the two metals shown, which are principally from automotive sources, correlate well with the oil and grease at the storm sewer outfalls, but not as well at combined sewer outfalls. This suggests that perhaps the oil and grease in stormwater sediments are primarily from petroleum origins. The geometric mean concentrations of oil and grease in combined sewer overflows and storm sewer discharges were 12.9 and 4.1 mg/l, respectively. Thus, one conclusion drawn from these results is that oil and grease are in greater concentrations in combined sewer overflows; however, they are better indicators of the presence of petroleum hydrocarbons in stormwater discharges.

Oil and grease concentrations from numerous urban stormwater and combined sewer sites across the nation are presented in Table 13. Also presented are the results from studies that measured specific petroleum hydrocarbons including a site where PAH's are measured. The overall reported ranges of oil and grease in urban stormwater and combined sewer overflows are shown in Figure 5. Maximum oil and grease concentrations in the urban stormwater have been reported to be greater than 100 mg/l while the CSO concentrations have been reported to be greater than 1000 mg/l.

Hunter et al. (30) measured urban stormwater petroleum hydrocarbon concentrations from a 1520 acre (616 ha) catchment (multi- and single-family residential land use) in Northern Philadelphia. The flow-weighted petroleum hydrocarbon concentrations for 5 storms varied from 2.2 to 5.3 mg/l. Peak concentrations were observed during the first portion of these storms. These peak concentrations were about 4 times greater than the flow-weighted storm averages. Aliphatic hydrocarbons represented slightly over two-thirds of the total petroleum hydrocarbons while the more toxic aromatic hydrocarbons represented the remainder. Interestingly, about 86 percent of the petroleum hydrocarbons was associated with particulates, which is encouraging since solids separation/removal is less costly and easier than removal of dissolved or fine colloidal matter.

A petroleum hydrocarbon mass balance estimation for Delaware Bay is shown in Table 14. Urban stormwater runoff was estimated to contribute significantly at about 10,500 lb/day (4760 kg/day) or 17 percent (30). However, after implementation of federally mandated effluent limits, urban stormwater runoff will contribute the greatest portion estimated at 39 percent of the total.

As mentioned earlier, Wakeham (28) investigated petroleum hydrocarbons in Lake Washington. Petroleum hydrocarbons in the sediments were found to

TABLE 10. POLYNUCLEAR AROMATIC HYDROCARBONS ON THE EPA PRIORITY POLLUTANT LIST.

Acenaphthene
Anthracene
Benz(a)anthracene
Benzo(b)fluoranthene
Benzo(k)fluoranthene
Benzo(a)pyrene
Benzo(ghi)perylene
Chrysene
Dibenz(a,h)anthracene
Fluoranthene
Fluorene
Indeno(1,2,3-c,d)pyrene
Naphthalene
Phenanthrene
Pyrene

TABLE 11. CONTRIBUTORS OF PETROLEUM HYDROCARBON TO THE OCEAN (27).

CONTRIBUTOR	PERCENT OF TOTAL*
Marine Transportation (Including Spills)	34
River Discharges**	26
Atmospheric Washoff	10
Natural Seeps	10
Industrial Wastes (Including Refineries)	8
Municipal Wastes	5
Urban Runoff	5
Offshore Production	2

*Total is approximately 6.1 million metric tons per year

**Assumed to also contain an urban runoff component

TABLE 12. LINEAR CORRELATION MATRIX (R^2) FOR OIL AND GREASE AND INDICATED PARAMETERS IN LAKE WASHINGTON SEDIMENTS (11).

Site	Pb	Zn	TOC	CH
Combined Sewer Outfalls	.491*	.093	.835**	.515
Storm Sewer Outfalls	.870**	.549**	.217	.821**
Controls	.173	.137	.035	.186

*95% Confidence Level

**99% Confidence Level

TOC = Total Organic Carbon
CH = Chlorinated Hydrocarbons

TABLE 13. PETROLEUM HYDROCARBONS AND HEM's IN URBAN WET-WEATHER DISCHARGES.

Parameter*	Average Conc. (mg/1)	Range (mg/1)	Type	Location	Ref.
Total HC	3.69	2.18 - 5.30	Urban Stormwater	Northern Philadelphia, Pa.	30
Aliphatic HC	2.57	1.50 - 3.50			
Aromatic HC	1.12	0.68 - 1.72			
Aliphatic HC	12	6 - 24	Highway/Bridge Runoff	Seattle, Washington	28
Aliphatic HC	1.2	0.2 - 7.5	Urban Stormwater		
Total HC	10	---	Highway Runoff	Sweden	29
Total HC	0.9	---	Urban Stormwater (multi-family)		
PAH's					
Fluoranthene	0.098	---	Urban Stormwater (Commercial)	Orlando, Florida	17
1,2-Benzanthracene	0.050	---			
Benzo(a)pyrene	0.170	---			
3,4-Benzofluoranthene	0.052	---			
Chrysene	0.050	---			
Anthracene	0.027	---			
1,12-Benzoperylene	0.041	---			
Indeno(1,2,3-c,d)pyrene	0.086	---			
Pyrene	0.076	---			
HEM	12.5	1.0 - .55.	Urban Stormwater (residential)	Seattle, Washington	2
HEM	20.0	0.5 - 40.	Urban Stormwater (residential)		
HEM	12.4	2.8 - 300.	Urban Stormwater (Industrial)		
HEM	11.5	1.0 - 130.	Urban Stormwater (Commercinl)		
HEM	9.5	0.2 - 52.	Urban Stormwater (residential)		
HEM	9.5	0.6 - 38.	Urban Stormwater (Low density residential)		
HEM	18.2	1.0 - 140.	Combined Sewer Overflow (Commercial)		
HEM	14.1	0.1 - 110.	Combined Sewer Overflow (Residential/Commercial)	San Francisco, California	2
HEM	17.8	0.8 -1327.	Combined Sewer Overflow (Residential/Industrial)		
HEM	34.4	3.1 - 423.	Combined Sewer Overflow (Residential/Commercial)		
HEM	11.9	0.4 - 122.	Combined Sewer Overflow (Residential)		
HEM	11.7	2.2 - 63.4	Combined Sewer Overflow (Residential/Commercial)		
HEM	12.5	6.6 - 21.5	Urban Stormwater (Residential/Commercial)		
HEM	7.5	4.5 - 17.9	Urban Stormwater (Residential/Commercial)		
HEM	---	1.3 - 64.	Combined Sewer Overflow	Rochester, New York	9
HEM	---	26. - 95.	Combined Sewer Overflow	New York City, New York	32
HEM	2.8	---	Urban Stormwater		
HEM	12.9	6.7 - 25.	Combined Sewer Overflow	Seattle, Washington	11
HEM	4.1	2.9 - 5.7	Urban Stormwater		

* HC = Hydrocarbons

HEM = Hexane Extractable Material

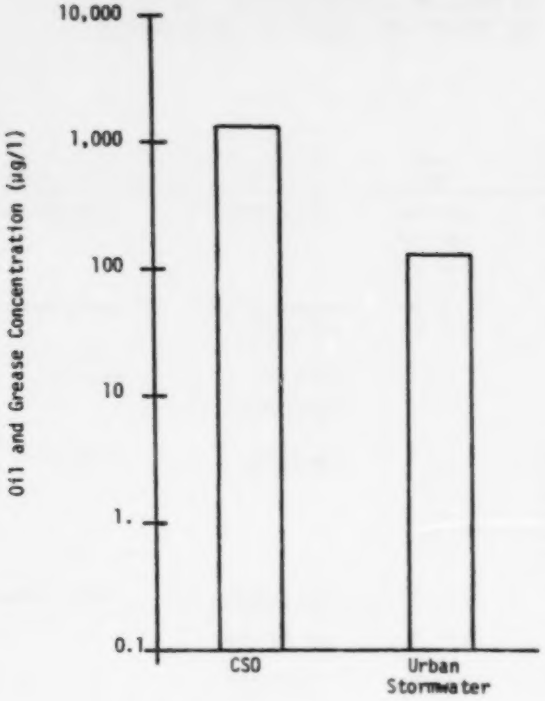


FIGURE 5. RANGES OF OIL AND GREASE IN URBAN WET-WEATHER DISCHARGES.

TABLE 14. PETROLEUM HYDROCARBON MASS BALANCE ESTIMATED TO DELAWARE BAY (30).

SOURCES	PRESENT		AFTER-IMPLEMENTING FEDERAL EFFLUENT LIMITS	
	Mass Load* (lb/day)	Percent	Mass Load* (lb/day)	Percent
SPILLS	6,000	9-10	6,000	23
POTW's	7,800-15,000	14-23	2,000	8
REFINERIES	24,000	37-42	1,900	7
OTHER INDUST.	8,900	14-16	6,200	23
URBAN RUNOFF	10,500	16-18	10,500	39

* lb = 0.454 kg

correlate with the growth history of the surrounding metropolitan area. Aliphatic hydrocarbons were sampled (grab samplets) in runoff from two heavily traveled bridges (i.e., 50,000 automobiles/day). Concentrations ranged from 6 to 24 mg/l with an average of about 12 mg/l. An urban stormwater runoff site had lower concentrations ranging from 0.2 to 7.5 mg/l. Primary wastewater treatment plant effluent samples were found to contain 3 to 5 mg/l of aliphatic hydrocarbons (wastewater treatment plant effluents no longer discharged to the lake; however, the lake still receives combined sewer overflows).

SUMMARY:

Urban stormwater discharges and combined sewer overflows have been shown to contribute large quantities of oxygen consuming material, heavy metals, and petroleum hydrocarbons which may adversely affect beneficial use of receiving waters. Of course, actual site studies should evaluate the impacts of urban wet-weather discharges considering other pollutant sources, receiving water characteristics, and economic and financial analyses before embarking on expensive control programs. Other pollutants should also be considered in the analysis. Investigations into the impact of oxygen consuming material from urban areas should consider the variable nature of both wet- and dry-weather discharges and the differences between the various wasteload characteristics (e.g., decay rates, reaeration rates, settleable oxygen consuming materials).

Several heavy metals have been shown to be in sufficient concentrations in urban stormwater discharges and combined sewer overflows to exceed water quality criteria for the protection of aquatic life even with many dilutions. Near outfalls (before dilution and mixing can occur) metals will cause biological stress areas as well. There are relatively strong indications that the major receiving water impacts from heavy metals will occur in the sediments. Load comparisons show that urban wet-weather discharges contribute more of several metals, such as cadmium, chromium and lead, to receiving waters than POTW effluents.

Urban stormwater and combined sewer overflows have been shown to contribute significant portions of oil and grease and petroleum hydrocarbons to the environment. Petroleum hydrocarbons, most notably PAH's, are known carcinogens; however, additional research is needed to determine the effects of chronic exposure at relatively low levels (compared to major spills) of petroleum hydrocarbons, particularly in the benthos. The contributions in urban stormwater runoff are roughly equal to those of municipal wastewater effluents. Together these sources account for at least 10 percent of the petroleum hydrocarbons entering the ocean and a greater percent to in-land and estuarine waters.

REFERENCES

1. Colston, N.V. Jr., "Characterization and Treatment of Urban Land Runoff," EPA-670/2-74-096, December 1974.

2. Huber, W.C., J.P. Heaney, K.J. Smolenyak, and D.A. Aggidis, "Urban Rainfall-Runoff-Quality Data Base: Update with Statistical Analysis," EPA-600/8-79-004, August 1979.

3. Lager, J.A., W.G. Smith, W.G. Lynard, R.M. Finn, and E.J. Finnemore, "Urban Stormwater Management and Technology: Update and User's Guide," EPA-600/8-77-014, September 1977.

4. Manning, M.J., R.H. Sullivan, and T.M. Kipp, "Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges: Volume III, Characterization of Discharges," EPA-600/2-77-064c, August 1977.

5. Heaney, J.P., W.C. Huber, M.A. Medina, Jr., M.P. Murphy, S.J. Nix, and S.M. Hasan, "Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges: Volume II, Cost Assessment and Impacts," EPA-600/2-77-064b, March 1977.

6. Keefer, T.N., R.K. Simons, and R.S. McQuivey, "Dissolved Oxygen Impact from Urban Storm Runoff," EPA-600/2-79-156, November 1979.

7. Environmental Protection Agency, "Federal Guidelines State and Local Pretreatment Programs, MCD-43," EPA-430/9-76-017a, January 1977.

8. Drehwing, F.J., A.J. Oliver, D.A. MacArthur, and P.E. Moffa, "Disinfection/Treatment of Combined Sewer Overflow, Syracuse, New York," EPA-600/2-79-134, August 1979.

9. Drehwing, F.J., C.B. Murphy, Jr., S.R. Garver, D.F. Geisser, and D. Bhargava, "Combined Sewer Overflow Abatement Program, Rochester, N.Y.: Volume II, Pilot Plant Evaluations," EPA-600/2-79-031b, July 1979.

10. Innerfeld,H., A. Forndran, D.D. Ruggiero, and T.J. Hartman, "Dual Process High-Rate Filtration of Raw Sanitary Sewage and Combined Sewer Overflows," EPA-600/2-79-015, March 1979.

11. Tomlinson, R.D., B.N. Bebee, A.A. Heyward, S.G. Munger, R.G. Swartz, S. Lazoff, D.E. Spyridakis, M.F. Shepard, R.M. Thom, K.K. Chew, and R.R. Whitney, "Fate and Effects of Sediments from Combined Sewer and Storm Drain Overflows in Seattle's Nearshore Waters," Draft Report EPA Grant No. R-805602, 1979.

12. Betson, R., "Urban Hydrology: A Systems Study of Knoxville, Tennessee," TVA, June 1976.

13. Black, Crow and Edisness, Inc., et al., "Non-Point Pollution Evaluation Atlanta Urban Area," Contract No. DACW 21-74-C-0103, May 1975.

14. Horkeby, B. and P. Malmquist, "Microsubstances in Urban Stormwater," International Symposium on the Effects of Urbanization and Industrialization on Hydrological Regime and on Water Quality, Amsterdam, October 1977.

15. Pitt, R., "Demonstration of Nonpoint Pollution Abatement Through Improved Street Cleaning Practices," EPA-600/2-79-161, August 1979.

16. Wanielista, M.P., Y.A. Yousef and J.S. Taylor, "Stormwater Management to Improve Lake Water Quality," Draft Report, EPA Grant No. R-805580, May 1980.

17. F.T. Brezenski, "Analytical Results - Orlando Florida," EPA Priority Pollutant Analysis Region II Laboratory, February 1980.

18. Environmental Protection Agency, "Water Quality Criteria, Request for Comments," Federal Register, Vol. 44, No. 62, March 15, 1979.

19. Environmental Protection Agency, "Water Quality Criteria: Availability," Federal Register, Vol.44, No. 144, July 25, 1979.

20. Gupta, M.K., E. Ballinger, S. Vanderah, C. Hansen, and M. Clark, "Handling and Disposal of Sludges from Combined Sewer Overflow Treatment: Phase I - Characterization," EPA-600/2-77-053a, May 1977.

21. Salotto, B.V., et al., "Elemental Analysis of Wastewater Sludges from 33 Wastewater Treatment Plants in the U.S.," EPA-902/9-74-002, May 1974.

22. Pisano, W.C., G.L. Aronson, C.S. Queiroz, F.C. Blanc, J.C. O'Shaughnessy, "Dry-Weather Deposition and Flushing of Combined Sewer Overflow Pollution Control," EPA-600/2-79-133, August 1979.

23. Bowan, H.J.M., Trace Elements in Biochemistry, Academic Press, New York, 1966.

24. Pitt, R. and M. Bozeman, "Water Quality and Biological Effects of Urban Runoff on Coyote Creek," First Phase Report (In Press), EPA Grant No. R-805418, June 1979.

25. Wilber, W.G. and J.V. Hunter, "The Impact of Urbanization on the Distribution of Heavy Metals in Bottom Sediments of the Saddle River," Water Resources Bulletin, Vol. 15, No. 3, 1979.

26. Grossling, B.F., "An Estimate of the Amounts of Oil Entering the Ocean," in Sources, Effects and Sinks of Hydrocarbons in the Aquatic Environment, American Institute of Biological Sciences, August 1976.

27. National Academy of Sciences, E.B. Wilson (Ed.), "Petroleum in the Marine Environment," Washington, D.C., 1975.

28. Wakeham, S.G., "A Characterization of the Sources of Petroleum Hydrocarbons in Lake Washington," Journal WPCF, Vol. 49, No. 7, 1977.

29. Hallhagen, A., "Survey of Present Knowledge and Discussion of Input of Petroleum to the Marine Environment in Sweden," Inputs, Fates and Effects of Petroleum in the Marine Environment Workshop, National Academy of Sciences, 1973.

30. Hunter, J.V., T. Sabatino, R. Gomperts, and M.J. MacKenzie, "Contribution of Urban Runoff to Hydrocarbon Pollution," Journal WPCF, Vol. 51, No. 8, 1979.

31. Feuerstein, D.L. and W.O. Maddaus, "Wastewater Management Program Jamaica Bay, New York: Volume I: Summary Report," EPA-600/2-76-222a, September 1976.

32. Shaheen, D.G., "Contributions of Urban Roadway Usage to Water Pollution," EPA-600/2-75-004, April 1975.

9

Eighth Session

STORMWATER MANAGEMENT

Moderator: Myron Tiemens
EPA, Washington, D.C.

THE USE OF RECEIVING WATER QUALITY MODELS IN URBAN RUNOFF POLLUTION ABATEMENT: APPLICATION TO MARGINAL BENEFIT - MARGINAL COST ANALYSIS

Cornelius B. Murphy, Jr.
Gregory J. Welter
Dwight A. MacArthur
O'Brien & Gere Engineers, Inc.
Syracuse, New York
and
Raymond P. Canale
University of Michigan
Ann Arbor, Michigan

Abstract

Urban storm runoff has been determined by a number of investigations to be a significant portion of the water pollution problem and abatement of this source has been recognized as a necessary consideration in achieving the national water quality goals of PL 92-500. However, due to the highly irregular nature of the runoff phenomenon, abatement measures that address this problem tend to be very expensive. Thus, in the EPA construction grants program great importance has been placed on careful planning demonstrating the cost-effectiveness of proposed projects (Program Guidance Memorandum 61).

The benefits of runoff pollution abatement measures must be evaluated in terms of projected receiving water quality conditions relative to some defined quality standards. There is considerable experience in the establishment of water quality standards and resultant effluent limitations for non-transient municipal and industrial discharges, generally through the use of mathematical models. Generally, these models are applied against a "design" receiving water hydrologic regime (for instance, 10-year, 7-day low flow) and effluent limitations selected which will meet the desired water quality standards. This steady-state analysis assumes a toleration of water quality contravention from the discharge of a low frequency corresponding to the designated low flow.

This approach is not wholly adequate to analysis of the problem posed by intermittent storm runoff discharges. These analyses require alternate modeling approaches and a restatement of pertinent standards to reflect the short-term high variability of the storm runoff event.

Using ongoing studies of combined sewer systems in Rochester, New York and Washington, D.C., as case studies, a procedure is presented for the application of receiving water analyses in urban runoff planning. The studies in both areas have included the development of water quality models based on familiar concepts of mass balance and calibrated against detailed field surveys and laboratory experiments.

In a Rochester study, steady state models of dissolved oxygen in the

Genesee and fecal coliform concentration in the Rochester Embayment of Lake Ontario are used to project receiving water conditions under dry-weather loads and an envelope of expected impacts of the combined sewer overflows under various system configurations. A time-variable model of Rock Creek and the Potomac and Anacostia Rivers has been used to project transient fecal coliform nutrient and dissovled oxygen concentrations resulting from overflows from the District of Columbia sewer system.

In an analysis of marginal costs and marginal benefits associated with various combined sewer overflow abatement alternatives, a series of model runs are made to project the impact of various waste discharges from the alternate system configurations on the receiving waters under several hydrologic regimes and ambient temperatures. The conjoint probabilities of the receiving water conditions and storm loads are determined on the basis of historical records and used to project the expected water quality under the alternative system configurations. These projections can be quantified as water quality improvements in terms of expected degree of contravention of stream standards with regards to frequency, duration, areal extent and peak concentration. On the basis of this analysis of project cost estimates drawn from generalized cost curves, marginal costs and benefits can be displayed graphically for each alternative.

The analysis of the Syracuse, New York, Rochester, New York and the District of Columbia combined sewer facilities planning activities are discussed relative to this marginal cost-marginal benefit approach as case studies.

Introduction

Urban storm runoff has been determined by a number of investigators to represent a significant portion of the water pollution problem and abatement of this source has been recognized as a necessary consideration in achieving the national water quality goals of PL 92-500. However, due to the highly irregular nature of the runoff phenomenon abatement measures that address this problem tend to be expensive and difficult to size. Thus, in the EPA construction grants program great importance has been placed on careful planning demonstrating the cost-effectiveness of proposed projects (PRM 75-34).

The benefits of runoff pollution abatement measures must be evaluated in terms of projected receiving water quality conditions relative to some defined quality standards. There is considerable experience in the establishment of water quality standards and resultant effluent limitations for non-transient municipal and industrial discharges, generally through the use of mathematical models. Frequently, these models are applied against a "design" receiving water hydrologic regime (for instance, 10-year, 7-day low flow) and effluent limitations are selected which will meet the desired water quality standards. This steady-state analysis assumes a toleration of water quality contravention for a low frequency discharge corresponding to flows less than Q7, 10.

This approach is not wholly adequate to analyze the problem posed by intermittent storm runoff discharges. These analyses require alternate modeling approaches and a re-statement of pertinent standards to reflect the short-term, high variability of the storm runoff event.

Using ongoing studies of combined sewer systems in Rochester, New York and Washington, D.C. as case studies, a procedure is presented for the application of receiving water analyses in urban runoff planning. The studies in both areas have included the development of water quality models based on familiar concepts of mass balance and calibrated against detailed field surveys and laboratory experiments. In the Rochester study, steady state models of dissolved oxygen in the Genesee River and fecal coliform concentrations in the Rochester Embayment of Lake Ontario are used to project receiving water conditions under dry-weather loads and an envelope of expected impacts of the combined sewer overflows under various system configurations. Results of the dissolved oxygen model for the Genesee River and fecal coliform model for the Rochester Embayment of Lake Ontario are presented. Time-variable models of Rock Creek and the Potomac and Anacostia Rivers are being developed to project transient fecal coliform, algal nutrient, and dissolved oxygen concentrations resulting from overflows from the District of Columbia sewer system.

In an analysis of marginal costs and marginal benefits associated with various combined sewer overflow abatement alternatives, a series of model runs are made to project the impact of various waste discharges from the alternate system configurations on the receiving waters under several hydrologic regimes and ambient temperatures. The joint probabilities of the receiving water conditions and storm loads are determined on the basis of historical records and are used to project the expected water quality under the alternative system configurations. These projections can be quantified as water quality improvements in terms of expected degree of contravention of stream standards with regards to frequency, duration, areal extent, and peak concentration. On the basis of this analysis of project cost estimates drawn from generalized cost curves, marginal costs and benefits can be displayed graphically for each alternative.

Case Study
Rochester, New York

As part of the Combined Sewer Overflow Facility Planning Activity conducted for the Monroe County Division of Pure Waters, O'Brien & Gere Engineers, Inc. and Limno-Tech, Inc. conducted a water quality evaluation program design to:

A. Establish the impact of existing combined sewer overflow pollutional loads on the Genesee River and the Rochester Embayment of Lake Ontario.

B. Project the effect of improvements to the existing combined sewer system on the receiving waters.

The combined sewer system of the Rochester Pure Waters District serves nearly 65% of the city, with overflows which combined into 15 discharges to the Genesee River, two discharges to Irondequoit Bay and one discharge to the New York State Barge Canal (Figure 1). An overflow monitoring and sampling program was initiated in early 1973 which involved the measurement of flow and quality at 13 major overflow discharges and three collection system locations in conjunction with a rain gauge network of 12 stations. Table 1 presents the average constituent concentrations for 5 of the parameters.

From the flow data measured during the 1975 monitoring program it had been estimated that the average overflow event involves a discharge of 52.8 MG of CSO generated wastewater to the Genesee. Utilizing the analytical data presented in Table 1, it has been determined that an average summer overflow event results in a discharge of the following constituent loadings to the Genesee River (1):

Biochemical Oxygen Demand	64,920 pounds/storm
Total Suspended Solids	136,860 pounds/storm
Total Inorganic Phosphate	540 pounds/storm
Zinc	110 pounds/storm
Lead	90 pounds/storm
Cadmium	1.3 pounds/storm

Modeling the existing combined sewer network utilizing the Simplified Stormwater Model (SSM) projects that the combined sewer system will contribute 42.7 overflow events during an average year. Utilizing this frequency factor and the derived average storm loading, it is possible to estimate the following annual combined sewer loadings (1):

Biochemical Oxygen Demand	2,772,050 pounds/year
Total Suspended Solids	5,844,040 pounds/year
Total Inorganic Phosphate	22,850 pounds/year
Zinc	4,870 pounds/year
Lead	3,750 pounds/year
Cadmium	54 pounds/year

To compare the annual impact of the Rochester combined sewer system with the total load carried by the Genesee River, the following annualized load of the Genesee River is presented as follows:

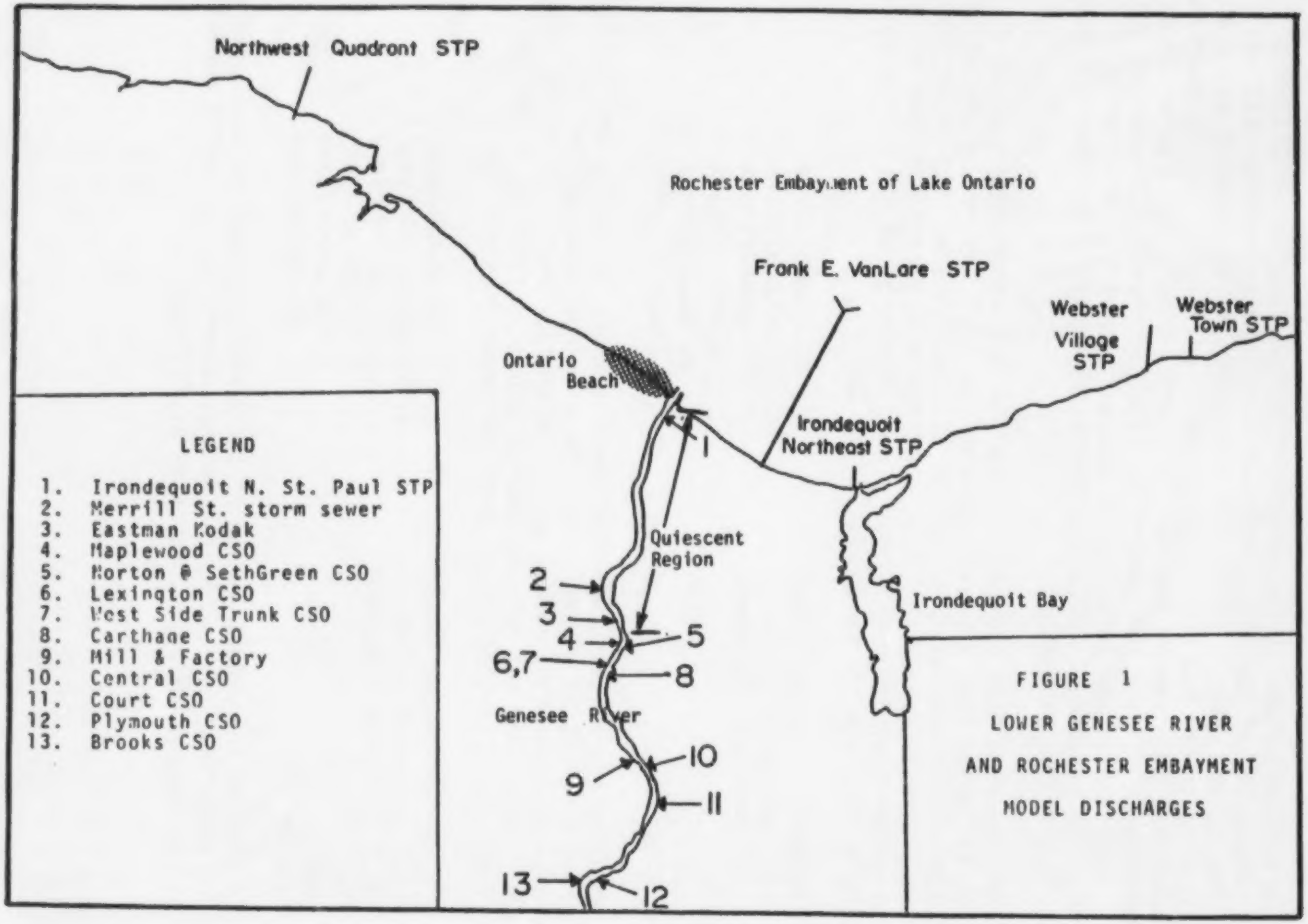


FIGURE 1
LOWER GENESEE RIVER
AND ROCHESTER EMBAYMENT
MODEL DISCHARGES

TABLE 1

1975

ROCHESTER COMBINED SEWER OVERFLOWS
AVERAGE CONCENTRATIONS

Area	BOD5	TOC	TSS	CR	CU	CD	PB	NI	ZN	TIP
7 Maplewood	161	77	317	0.02	0.09	0.004	0.2	0.02	0.27	1.42
8 Lexington	158	77	320	0.03	0.09	0.004	0.2	0.02	0.29	1.28
9 West Side	178	93	358	0.02	0.10	0.003	0.2	0.02	0.27	1.36
16 Mill and Factory	188	98	366	0.007	0.09	0.003	0.2	0.02	0.27	1.37
17 Plymouth	186	98	358	0.007	0.09	0.003	0.2	0.01	0.27	1.38
18 Brooks	183	99	354	0.007	0.09	0.003	0.2	0.01	0.27	1.37
21 Norton-Seth Green	203	111	381	0.008	0.09	0.003	0.2	0.01	0.28	1.34
22 Carthage	205	111	377	0.008	0.09	0.003	0.2	0.01	0.28	1.35
25 Central	53	41	226	0.006	0.08	0.003	0.2	0.02	0.24	1.04
26 Court	52	41	205	0.005	0.08	0.003	0.2	0.01	0.24	1.08
28 Screenhouse	60	46	168	0.005	0.07	0.002	0.2	0.02	0.22	0.50
29 Densmore By-Pass	No data available									
31 THomas Creek	67	53	292	0.008	0.11	0.002	0.09	0.004	0.27	2.85

NOTE: All values in mg/l

SOURCE: Unpublished data
Rochester Combined Sewer Overflow R and D Program
O'Brien & Gere Engineers, Inc.

Biochemical Oxygen Demand	5,271,700	pounds/year
Total Suspended Solids	683,606,680	pounds/year
Total Inorganic Phosphate	181,040	pounds/year
Zinc	91,980	pounds/year
Cadmium	11,680	pounds/year

The projections indicate that the combined sewer system contributes only a fraction of the annualized loading of the Genesee River to Lake Ontario for parameters measured except for Biochemical Oxygen Demand. Even though most of the loading projections are quite low, the CSO impact is of a very short and intense nature. This transient wet-weather impact creates significant adverse stress on the aquatic ecosystem which occurs fairly frequently on an annual basis. However, its mark is also left in a less intensive but still significant magnitude in the form of particulate pollutants contributed to the benthos in the quiescent, lower reaches of the Genesee River.

In light of this setting, it was determined that the likely transient water quality impacts induced in the lower reaches of the Genesee River involve short-term dissolved oxygen and fecal coliform water quality violations. The most likely transient water quality contravention within the Rochester Embayment involves elevated fecal coliform levels in the vicinity of near shore beach areas. This is supported by an incomplete historical water quality data base which illustrates a condition of variable and elevated indicator organism densities in both the Genesee River and Rochester Embayment as well as frequently depressed dissolved oxygen concentrations within the water column of the lower reaches of the Genesee River.

The seven historical samplings of the sanitary conditions of the Genesee River dating back as early as 1912 support the initial water quality impact assessment. However, the historical data are not adequate with respect to spatial detail and lack associated discharge information needed to make a quantified water quality assessment. Therefore,

O'Brien & Gere Engineers performed four extensive water quality surveys during 1975, two of which were during storm events (2).

The impact of combined sewer overflow discharges on the water quality of the Genesee River and the Rochester Embayment of Lake Ontario is illustrated by the water quality data collected during the November 10-12, 1975 wet-weather survey. The overflow event was initiated by rainfall of 0.38 inches over a span of 8 hours and was estimated to result in peak CSO discharge rates of 154 cfs, receiving water flows of 2300 cfs and water column temperatures of 13°C.

Figures 2-4 illustrate the minimum, average and maximum water column parameter concentrations at each of the sampling stations for the duration of the survey. As an aid in the interpretation of the water column response data, one should realize that the significant CSO discharges are located along the region from milepoint 5.3 to milepoint 10.7.

Genesee River Models

The wet and dry-weather data sets were used to calibrate and verify three water quality models developed for the Genesee River and the Rochester Embayment. These involved steady state dissolved oxygen and fecal coliform models for the Genesee River and a steady state fecal coliform model for the Rochester Embayment of Lake Ontario. The basic model framework used to simulate water quality in the Genesee River was a Limno-Tech developed modification of the U.S. Environmental Protection Agency steady state AUTO-QUAL modeling system. The model produces a steady state uni-directional description of rivers using second-order, finite difference approximations. The structure of the model is based on the simple concepts of continuity, including terms for advection, dispersion, reaction, and point or non-point sources or sinks. The model can calculate the steady-state spatial distribution of any water quality parameter whose kinetic behavior can be described by zero or first-order kinetic formulations or described as conservative. In the case of the Genesee River, the parameters modeled included chloride, fecal coliform, carbonaceous oxygen demand, nitrogenous oxygen demand, and dissolved oxygen. Details concerning the model framework are presented in Table 2.

Rochester Embayment Model

A review of the historical data base developed on the Rochester Embayment of Lake Ontario indicated the necessity to collect wet-weather field data to serve as the basis for developing a fecal coliform model. Fecal coliform was selected as the water quality reference parameter because of a long history of variable sanitary conditions exhibited at existing and potential beach areas within the Embayment.

Because extensive data did not exist for the Embayment Fecal Coliform model, a three day investigation of coliform and chloride conditions in the Embayment was conducted. During a 72 hour period following the November 10, 1975 storm, O'Brien & Gere Engineers, Inc. monitored the water quality at ten stations as well as all known sources of Embayment contamination. Figures 5-8 present the fecal coliform and chloride data measured on November 11th and 12th. The total measured loading to the Rochester Embayment from the Genesee River for November 11th and November 12th are 2.48×10^{14} cells/day and 7.84×10^{13} cells/day, respectively.

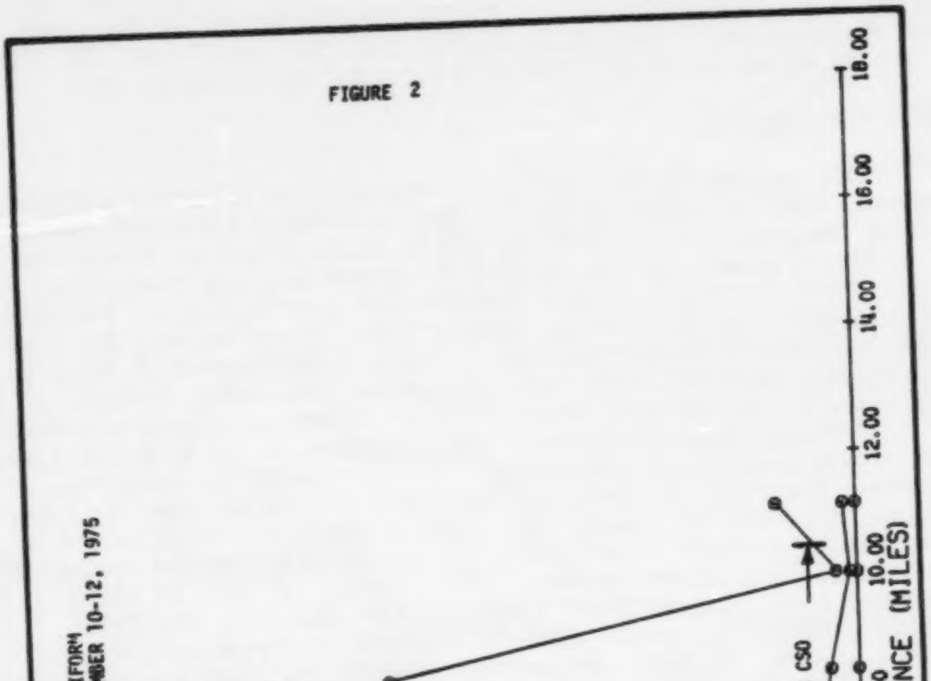
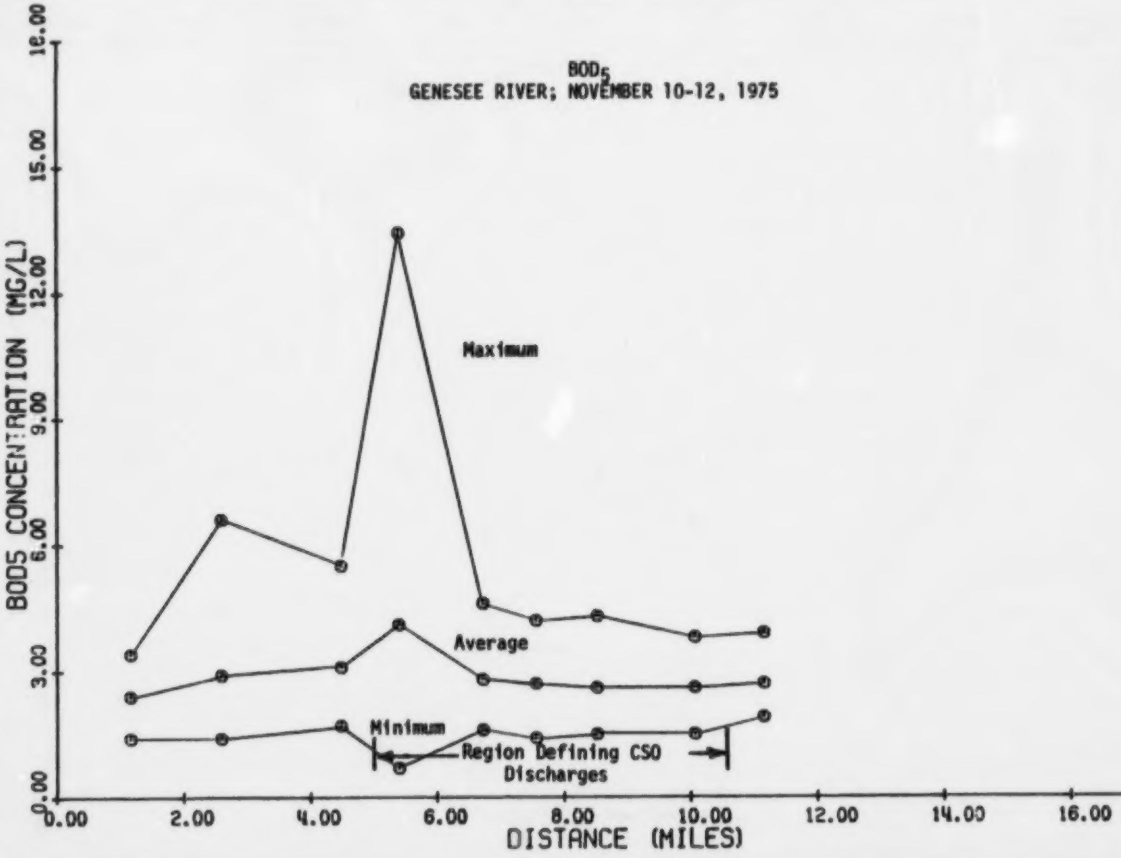


FIGURE 2

FECAL COLIFORM
GENESEE RIVER; NOVEMBER 10-12, 1975

FIGURE 3



BOD_5
GENESEE RIVER; NOVEMBER 10-12, 1975

FIGURE 4

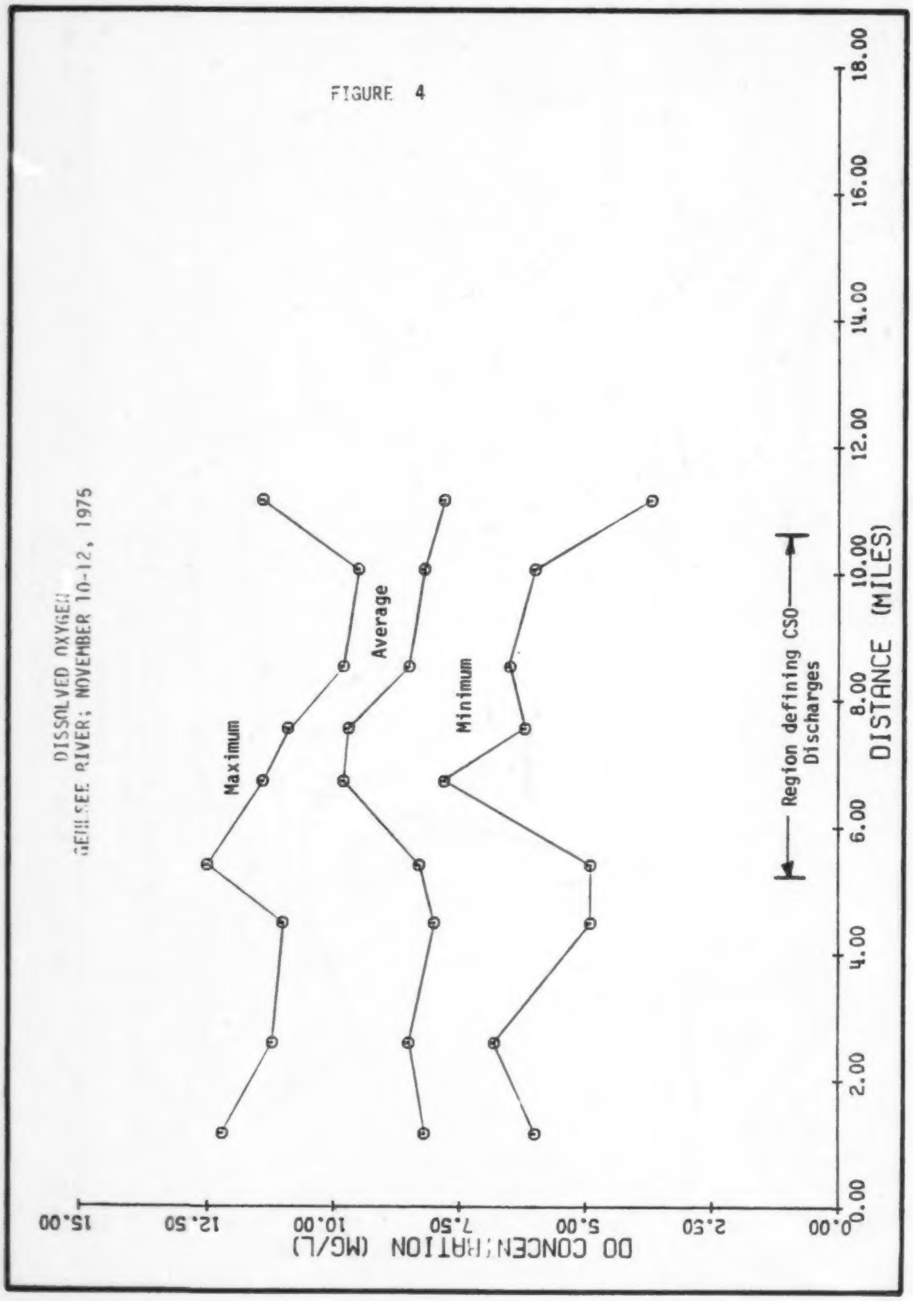

DISSOLVED OXYGEN
GENESEE RIVER; NOVEMBER 10-12, 1975
Maximum
Average
Minimum
Region defining CSO Discharges
DO CONCENTRATION (MG/L)
0.00
2.50
5.00
7.50
10.00
12.50
15.00
DISTANCE (MILES)
0.00
2.00
4.00
6.00
8.00
10.00
12.00
14.00
16.00
18.00

TABLE 2

GENESEE RIVER WATER QUALITY MODEL TECHNICAL INFORMATION

Type: Steady State

Predictive Capability: Chloride, Fecal Coliform, Carbonaceous Oxygen Demand, Nitrogenous Oxygen Demand, and Dissolved Oxygen

Basic Model Framework:

$$\frac{\partial c}{\partial t} = 0 = \frac{1}{A} \frac{\partial}{\partial x} AE\frac{\partial c}{\partial x} - \frac{1}{A} \frac{\partial}{\partial x} (Qc) + r$$

Dissolved Oxygen (DO) Differential Equation

$$\frac{\partial DO}{\partial t} = 0 = \frac{1}{A} \frac{\partial}{\partial x} AE \frac{\partial DO}{\partial x} - \frac{1}{A} \frac{\partial}{\partial} (QDO) - K_{rc}CBOD - K_{rn}NBOD$$

$$+ K_a(DO_s - DO) + \frac{1}{H} (P-R-B)$$

Fecal Coliform (N) Differential Equation

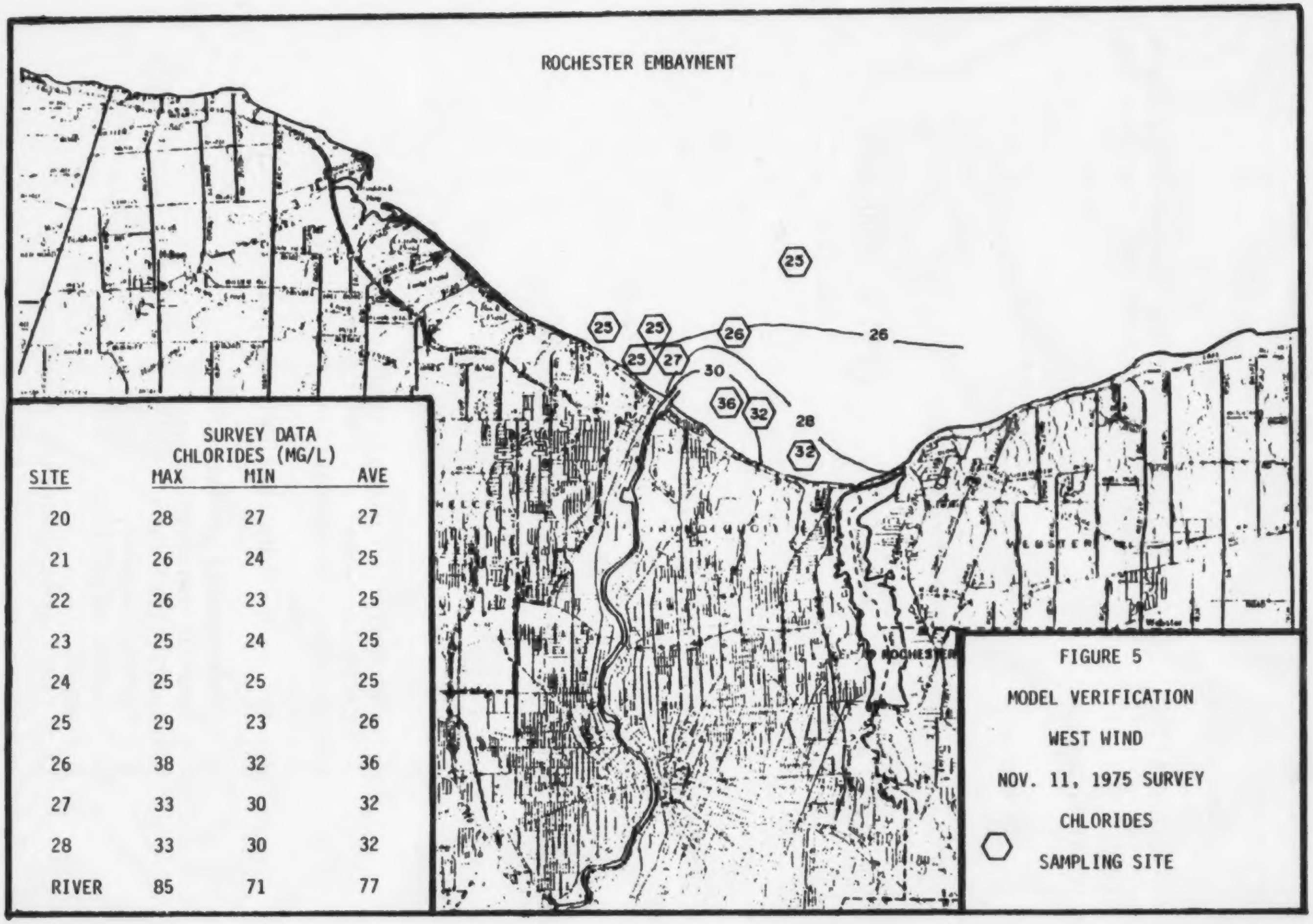
$$\frac{\partial N}{\partial t} = 0 = \frac{1}{A} \frac{\partial}{\partial x} AE\frac{\partial N}{\partial x} - \frac{1}{A} \frac{\partial}{\partial x} (QN) - K_f(N)$$

Conservative Substance (C) Differential Equation

$$\frac{\partial c}{\partial t} = 0 = \frac{1}{A} \frac{\partial}{\partial x} AE\frac{\partial c}{\partial x} - \frac{1}{A} \frac{\partial c}{\partial x} (Qc)$$

where

c = concentration of pollutant
A = cross sectional area of channel
E = longitudinal dispersion coefficient
Q = flow rate
r = net rate of biochemical reaction
t = time
x = longitudinal distance
P = rate of photosynthesis
R = respiration rate
B = benthal demand
DO_s = oxygen solubility
H = depth
Ki = first order reaction rates



SURVEY DATA
CHLORIDES (MG/L)

SITE	MAX	MIN	AVE
20	28	27	27
21	26	24	25
22	26	23	25
23	25	24	25
24	25	25	25
25	29	23	26
26	38	32	36
27	33	30	32
28	33	30	32
RIVER	85	71	77

FIGURE 5
MODEL VERIFICATION
WEST WIND
NOV. 11, 1975 SURVEY
CHLORIDES

⬡ SAMPLING SITE

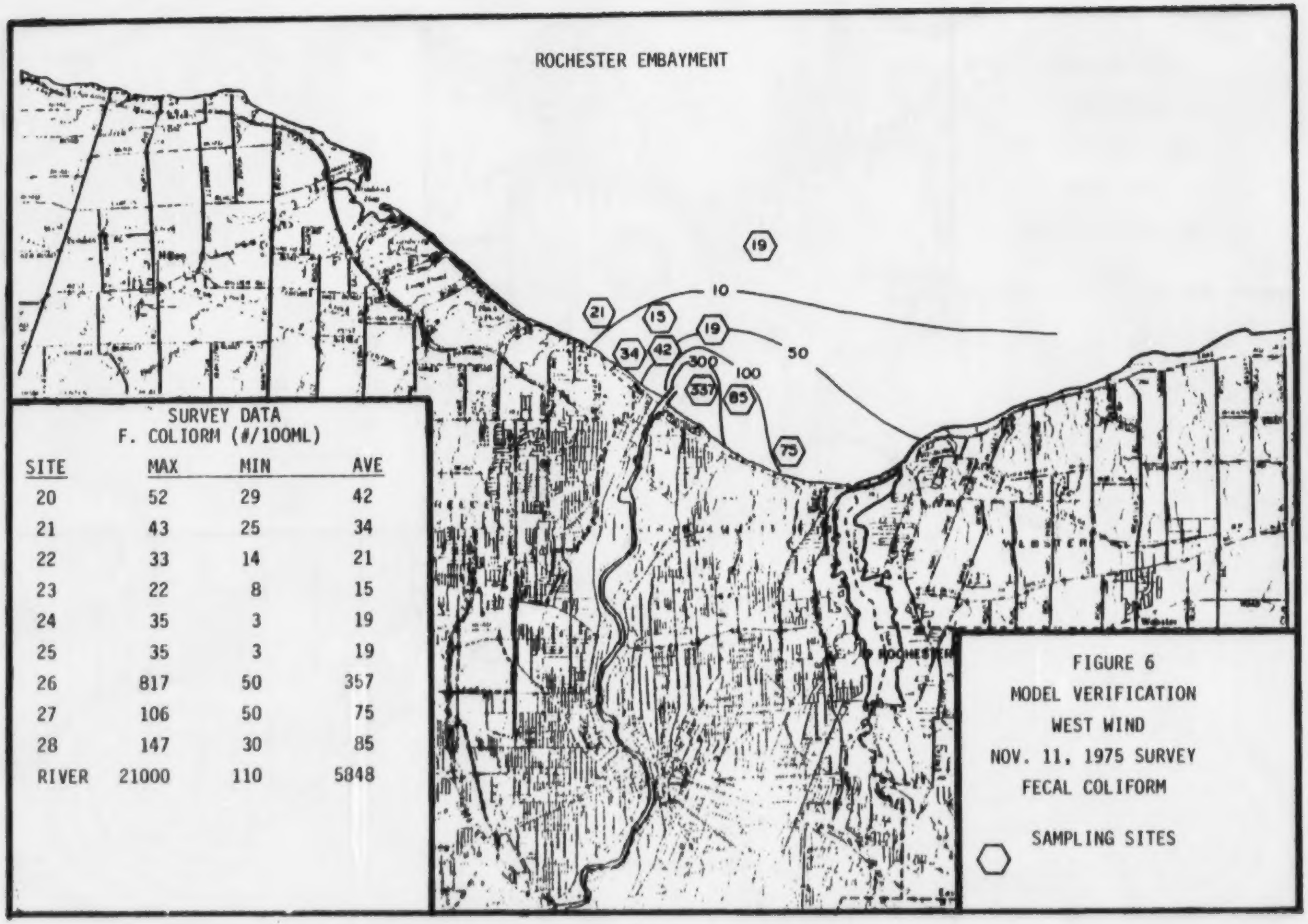
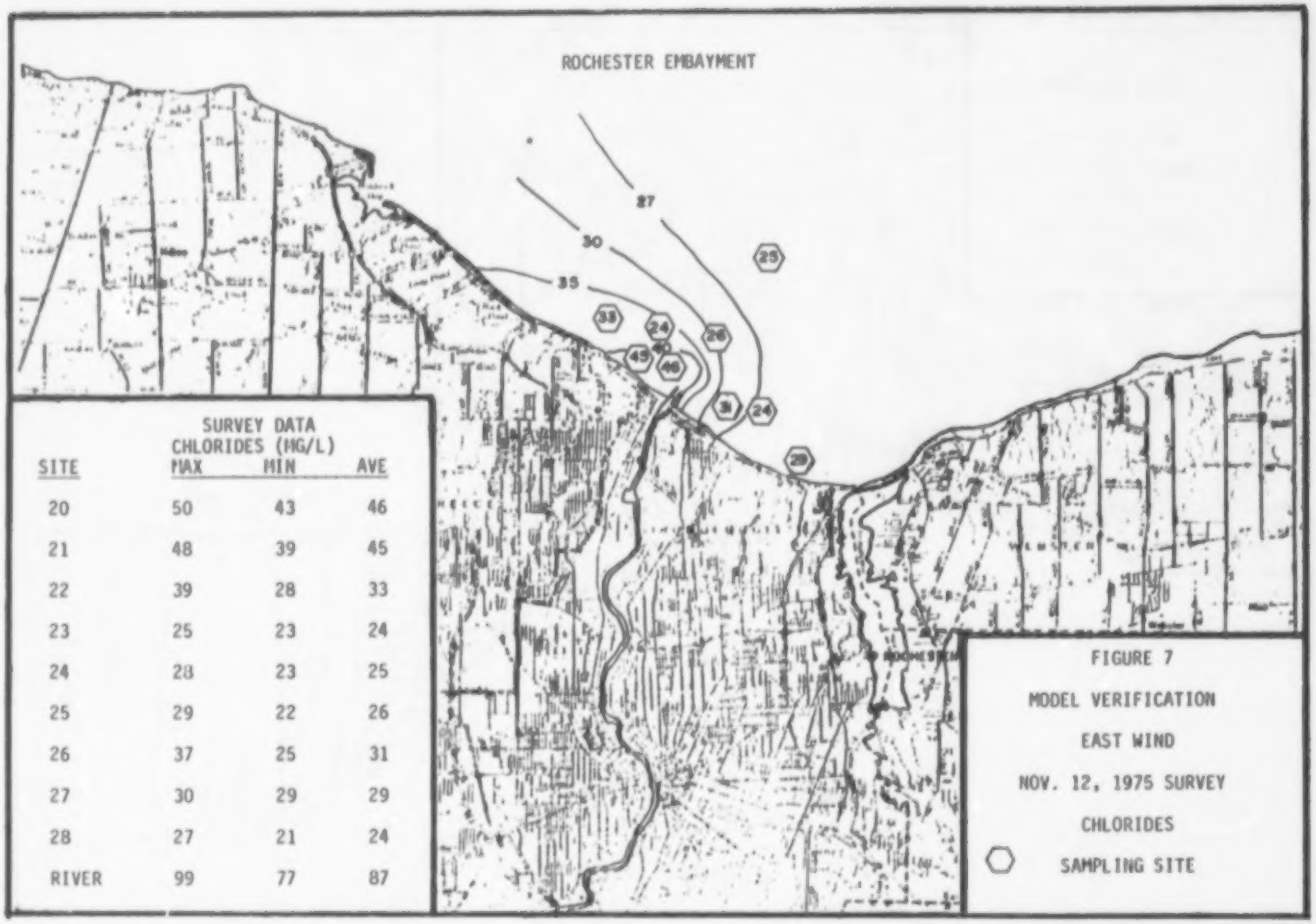


FIGURE 6
MODEL VERIFICATION
WEST WIND
NOV. 11, 1975 SURVEY
FECAL COLIFORM

⬡ SAMPLING SITES

SURVEY DATA
F. COLIORM (#/100ML)

SITE	MAX	MIN	AVE
20	52	29	42
21	43	25	34
22	33	14	21
23	22	8	15
24	35	3	19
25	35	3	19
26	817	50	357
27	106	50	75
28	147	30	85
RIVER	21000	110	5848



SITE	SURVEY DATA CHLORIDES (MG/L) MAX	MIN	AVE
20	50	43	46
21	48	39	45
22	39	28	33
23	25	23	24
24	28	23	25
25	29	22	26
26	37	25	31
27	30	29	29
28	27	21	24
RIVER	99	77	87

FIGURE 7

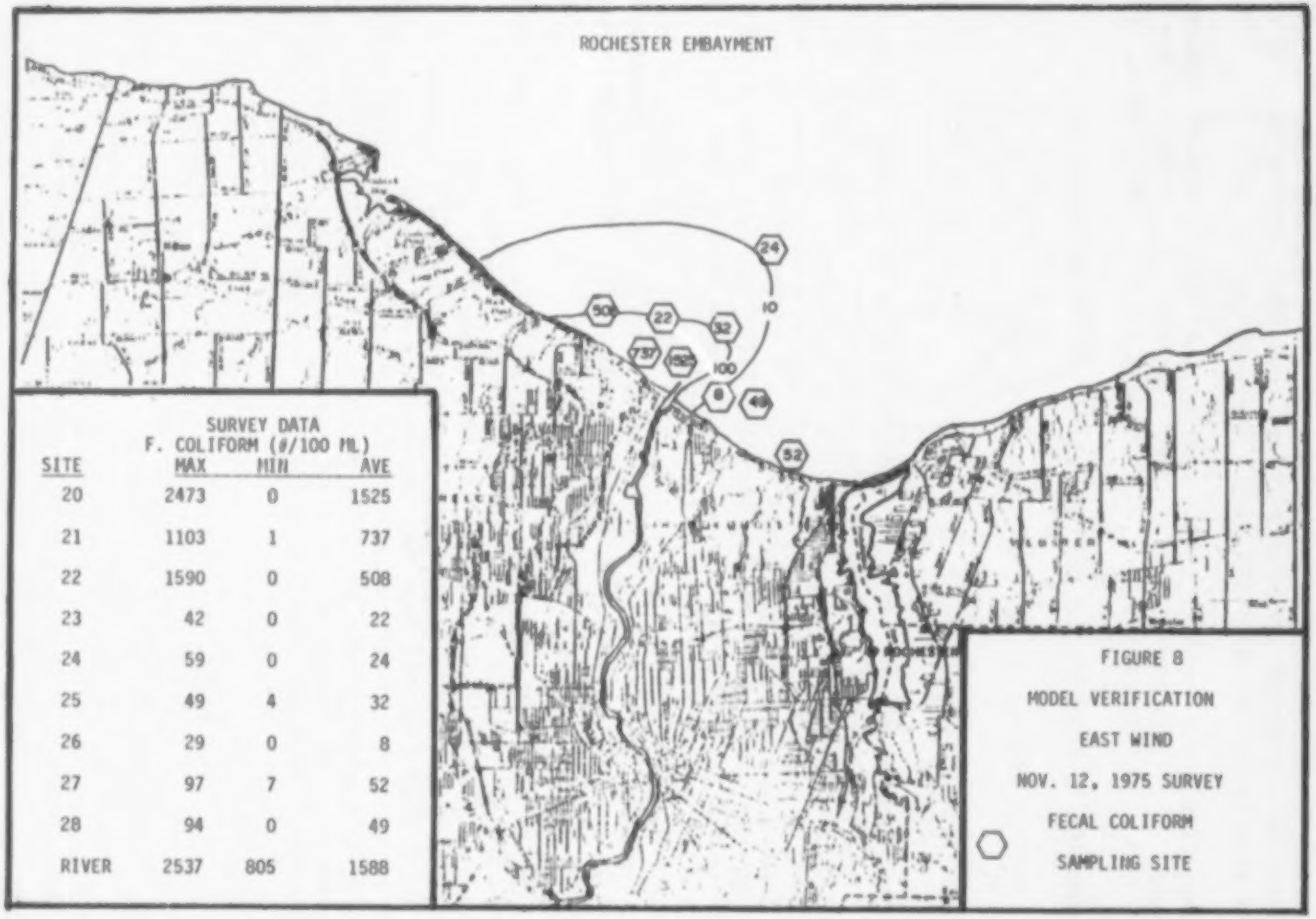
MODEL VERIFICATION

EAST WIND

NOV. 12, 1975 SURVEY

CHLORIDES

⬡ SAMPLING SITE



SURVEY DATA
F. COLIFORM (#/100 ML)

SITE	MAX	MIN	AVE
20	2473	0	1525
21	1103	1	737
22	1590	0	508
23	42	0	22
24	59	0	24
25	49	4	32
26	29	0	8
27	97	7	52
28	94	0	49
RIVER	2537	805	1588

FIGURE 8

MODEL VERIFICATION

EAST WIND

NOV. 12, 1975 SURVEY

FECAL COLIFORM

⬡ SAMPLING SITE

The conceptual basis for the embayment model is very similar to that of the Genesee River model. The concept of conservation of mass is intrinsic in the model. The two mechanisms which determine the fate of material in the system are transport and reaction. For the general case these effects can be described by the equation:

$$\begin{array}{llllllll} \text{accumulation} & = & \text{net input of mass} & + & \text{net input of mass} & + & \text{rate of disappearance} \\ \text{of mass} & & \text{from convective} & & \text{from dispersional} & & \text{or appearance of} \\ & & \text{transport} & & \text{transport} & & \text{mass due to reaction} \end{array}$$

Mathematically the continuity of material can be expressed as:

$$\frac{\partial c}{\partial t} = \nabla \cdot (E \nabla c) - \nabla (uc) + r$$

where u is the velocity

The embayment model differs from the Genesee River model in that the transport terms for the embayment necessitate a three dimensional description. Therefore it was necessary to approximate the continuous system with a system of completely mixed segments which allow for advective and dispersive flow among adjacent cells. A system of 132 cells was used to describe the embayment. The model has 87 surface segments which describe the lake to a depth of 20 meters and 45 bottom layer cells which describe all waters below the 20 meter depth contour. Six additional one-mile segments for the lower reaches of the Genesee River were included in the embayment model to adequately handle river-embayment interactions from upstream loads to the river. The modeling techniques used are similar to those described in reference (3).

One of the most important steps in developing a three dimensional water quality model is the identification and quantification of the hydraulic circulation within the system of study. A report prepared by Bonham-Carter, Thomas, and Lockner (4) which was generated as part of the International Field Year for the Great Lakes investigation on Lake Ontario presents a numerical model which calculates three dimensional, steady-state water transport on the embayment under unstratified conditions. The results of the IFYGL circulation model were not available in numerical form but rather were depicted in graphical vector form.

Vector analysis was used to transform the graphical information into a numerical scheme suitable for use in the water quality model. Circulation pattern predictions were available for steady winds from the north, south, east, west, northwest and southwest directions.

The development of the embayment fecal coliform model required an accurate quantification of the die-off coefficient. This information was derived from laboratory bacterial die-off studies performed specific to the Genesee River -Rochester·Embayment system. The decay coefficients used in model simulation were consistent with the ambient water temperature. For projections a die-off coefficient of 1.3 day^{-1} was used for warm water conditions.

The steady state embayment model was verified against four sets of coliform data (July 18, 1912; August 2, 1912; November 11, 1975, and November 12, 1975) and two sets of chloride data (November 11, 1975 and November 12, 1975). Model verification of the chloride data is useful because chloride is considered to be a conservative substance and hence provides an excellent means for testing the model's advective and dispersive transport terms without the complication of reaction processes. Figures 5-8 show the chloride and fecal coliform model projections compared to the actual data for the November 11, 1975 and November 12, 1975 data sets.

It is obvious that steady state models for the River and Bay cannot accurately describe time variable changes in water quality due to overflows of combined sewage. Therefore model calibration is qualitative. However, steady state models can be used to calculate the maximum response expected from CSOs (assuming steady maximum loads). Therefore steady state models can be used to compare and select planning and management alternatives for CSOs. The next section which deals with similar problems in the Washington DC area will utilize dynamic models which are calibarted under wet weather conditions.

Abatement Alternative Analyses

An extensive sensitivity analysis was conducted in an effort to define the critical receiving water conditions which result in the simulation of maximum CSO impact. The following represents a listing of parameters for which the sensitivity analysis was conducted:

Genesee River Fecal Coliform Model
- receiving water flows
- water column temperatures
- dispersion rates
- discharge location
- fecal coliform loading

Genesee River Dissolved Oxygen Model
- receiving water flows
- water column temperatures
- dispersion rates
- reaeration rates
- deoxygenation rates
- BODc and BODN loading
- benthic oxygen demand rates

Rochester Embayment Fecal Coliform Model
- wind direction and intensity
- water column temperatures
- fecal coliform decay rates
- fecal coliform loading

As a result of conducting the sensitivity analysis, it became quite apparent that the receiving water impact was as dependent on the pre-event water column conditions as it was on the actual CSO loading. Most of the significant determining variables for the Genesee River fecal coliform and dissolved oxygen models are related to the seasonal conditions generally characteristic of the Genesee River. Similarly, the frequency and magnitude of CSO discharges is statistically related to the seasonal reference.

Figure 9 presents the Genesee River critical dissolved oxygen projection for a single treatment plant treating at various combined sewer treatment rates and different treatment efficiencies. From this analysis it can be seen that only 25 MGD of CSO treated at an 85% removal efficiency may be discharged to the Genesee River under the analyses for critical low flow conditions and maximum water column temperatures without violating the water column dissolved oxygen level of 4.0 mg/l. Figure 10 presents the same CSO loading and treatment efficiency response under average summer flows. Under average summer flows 25 MGD of untreated CSO may be discharged without any treatment and still a dissolved oxygen concentration of 4.0 mg/l will be maintained.

As a result of these analyses and average summer Genesee River flow vs. loading analysis it was concluded that the combined sewer overflow treated discharge should be directed to the Rochester Embayment of Lake Ontario via conveyance to the Frank E. Van Lare sewage treatment plant for treatment utilizing existing and supplemental treatment facilities. The question that remained involved the sizing of the required conveyance and in-line storage facilities.

The resulting benefit framework became the frequency of dissolved oxygen violations in the Genesee River and the frequency of beach closing days projected for Ontario Beach which is located on the Rochester Embayment of Lake Ontario.

Six structurally intensive alternatives and a series of source and collection system management Best Management Practice abatement alternatives were evaluated in the course of developing a Master Plan for the Rochester combined sewer system. These alternatives are outlined as follows:

Figure 9

CRITICAL DO SAG PROJECTIONS

Alternative: Single Treatment Plant

Flow Regime: MA7CD/10

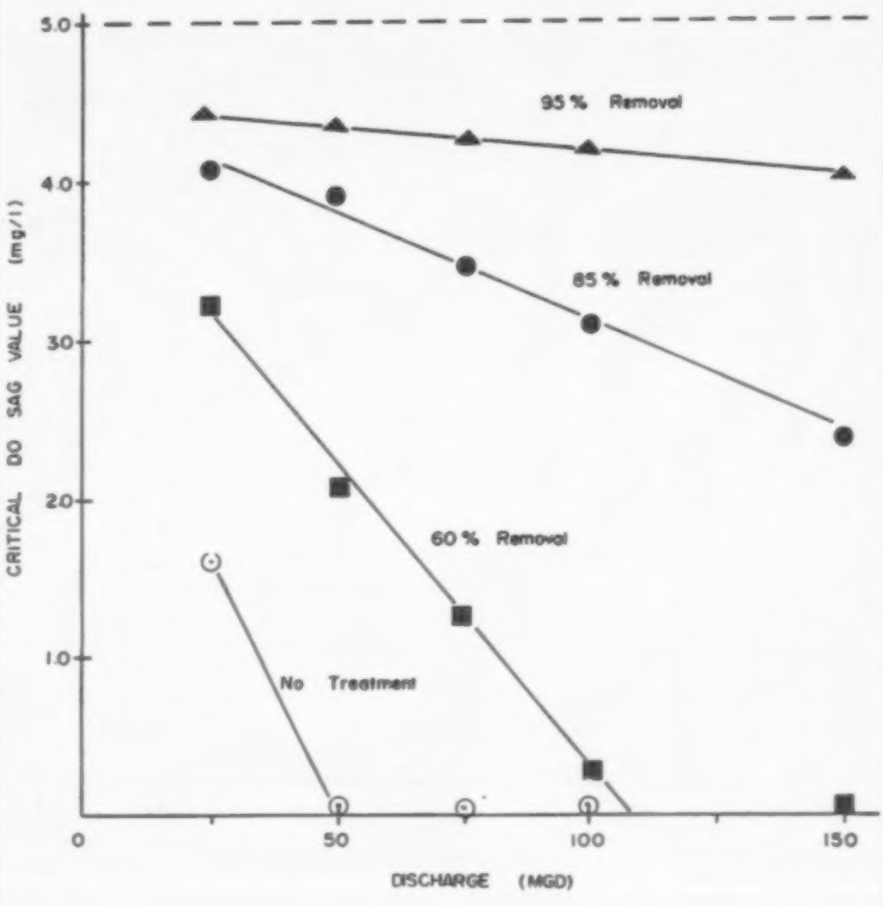
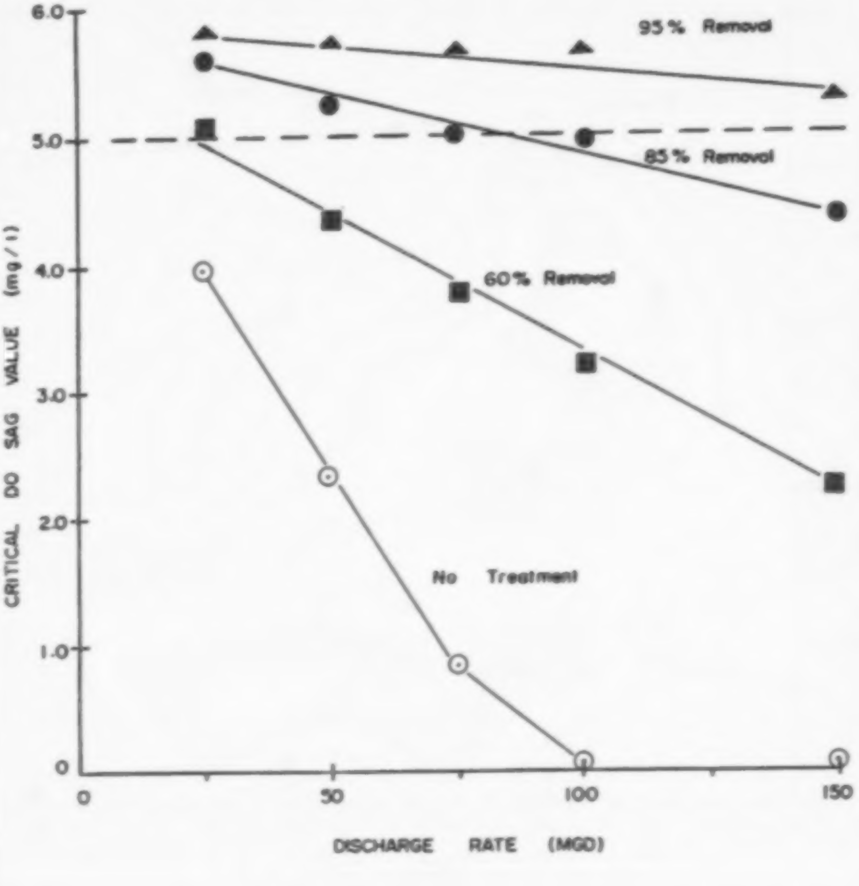


Figure 10

CRITICAL DO SAG PROJECTIONS

Alternative: Single Treatment Plant
Flow Regime: Average Summer



Structurally Intensive Alternatives

1. Capture of first flush from the River overflow sites and treatment of all post first-flush flows with primary swirl devices.

2. Various storage and treatment options for capturing and treating the total overflow to the Genesee River with treatment facilities located on the Genesee River.

3. Various storage and treatment options for capturing and treating the total overflow to the Genesee River with treatment facilities located at the present Van Lare STP and discharge to the Rochester Embayment of Lake Ontario.

4. Capture of first flush from the River overflow sites and discharge of untreated post first flush flows.

5. Use of primary swirl devices on each of the River overflow locations for treatment of the entire overflow volume; the swirl effluents being discharged directly to the Genesee River.

6. Storage and treatment balance for intercepting the River overflows and conveyance of the overflows to the Cross-Irondequoit Tunnel for treatment at the Van Lare STP.

Best Management Practice Options

1. Interceptor improvements to improve the conveyance capacity of the existing St. Paul Blvd. Interceptor.

2. Regulator modifications to optimize the conveyance of wet-weather flows utilizing the upgraded interceptor capacity.

3. Selective regulation of high impacting overflows.

4. Implementation of control structures to optimize the utilization of the available in-system storage capacity.

5. Optimal utilization of existing street sweeping and sewer cleaning operations.

6. Modification of the F.E. Van Lare existing treatment to allow split flow operation. Under wet-weather conditions, flows in excess of the process capacity of the secondary facilities receive chemical assisted primary treatment and are blended with the secondary treated flow prior to discharge.

On the basis of either water quality limitations or a cost-effectiveness analysis, the structurally intensive alternatives presented as 1, 2, 4, 5 or 6 were eliminated from further consideration. Structurally intensive alternative 3 was selected for further consideration. A marginal benefit/ marginal cost analysis was conducted on alternative 3. The EPA developed SWMM II model and the Dorsch Consult model were utilized to project the hydraulic loading from the existing system and the storage and treatment balance projected for various storage capacities. Figure 11 presents the number of Genesee River dissolved oxygen contraventions as a function of the effective in-line storage capacity.

In the development of the Master Plan for the Rochester combined sewer system the pertinent Best Management Practice alternatives have been identified for implementation followed by the implementation of the structurally intensive alternative as developed from the net a C/B D Ω. contravention analysis presented in Figure 11. The reduction in the number of beach closing days at the Ontario Beach and the number of dissolved oxygen contraventions encountered in the Genesee River were plotted as a function of program implementation. The improvement in water quality resulting from implementation of the Master Plan is presented in the form of Figures 12 and 13.

Case Study
Washington, D.C.

As part of the Phase I Combined Sewer Overflow being conducted on the Potomac-Anacostia River System for the Government of the District of Columbia, receiving water quality investigations were conducted to provide a basis for assessing the benefits of any modifications of the structural or operating characteristics of the combined sewer system proposed for the purpose of CSO pollution abatement. The water quality investigations in this study were focused on Rock Creek, Potomac River, and Anacostia River all of which receive significant CSO discharges from the District. Figure 14 illustrates the study area, the areas of significant CSO discharges and the park land or recreational space adjoining the water front.

Reconnaissance surveys conducted during the Phase I program included both dry and wet-weather water quality sampling and sediment analysis in the Potomac and Anacostia Rivers and Rock Creek. The water quality of Rock Creek was sampled for dissolved oxygen, fecal coliform, and suspended solids thirteen times during dry weather between July and October 1978 and intensively once subsequent to a rain event. Rock Creek sediments were sampled once for concentrations of selected heavy metals and chlorinated pesticides, and once to assess the macroinvertebrate community. The Potomac and Anacostia Rivers were surveyed once during wet weather conditions for dissolved oxygen, fecal coliform and suspended solids, and also for sediment oxygen demand and sediment concentrations of selected heavy metals, chlorinated pesticides, and oil and grease.

FIGURE II
DISSOLVED OXYGEN
CONTRAVENTION REDUCTION
GENESEE RIVER

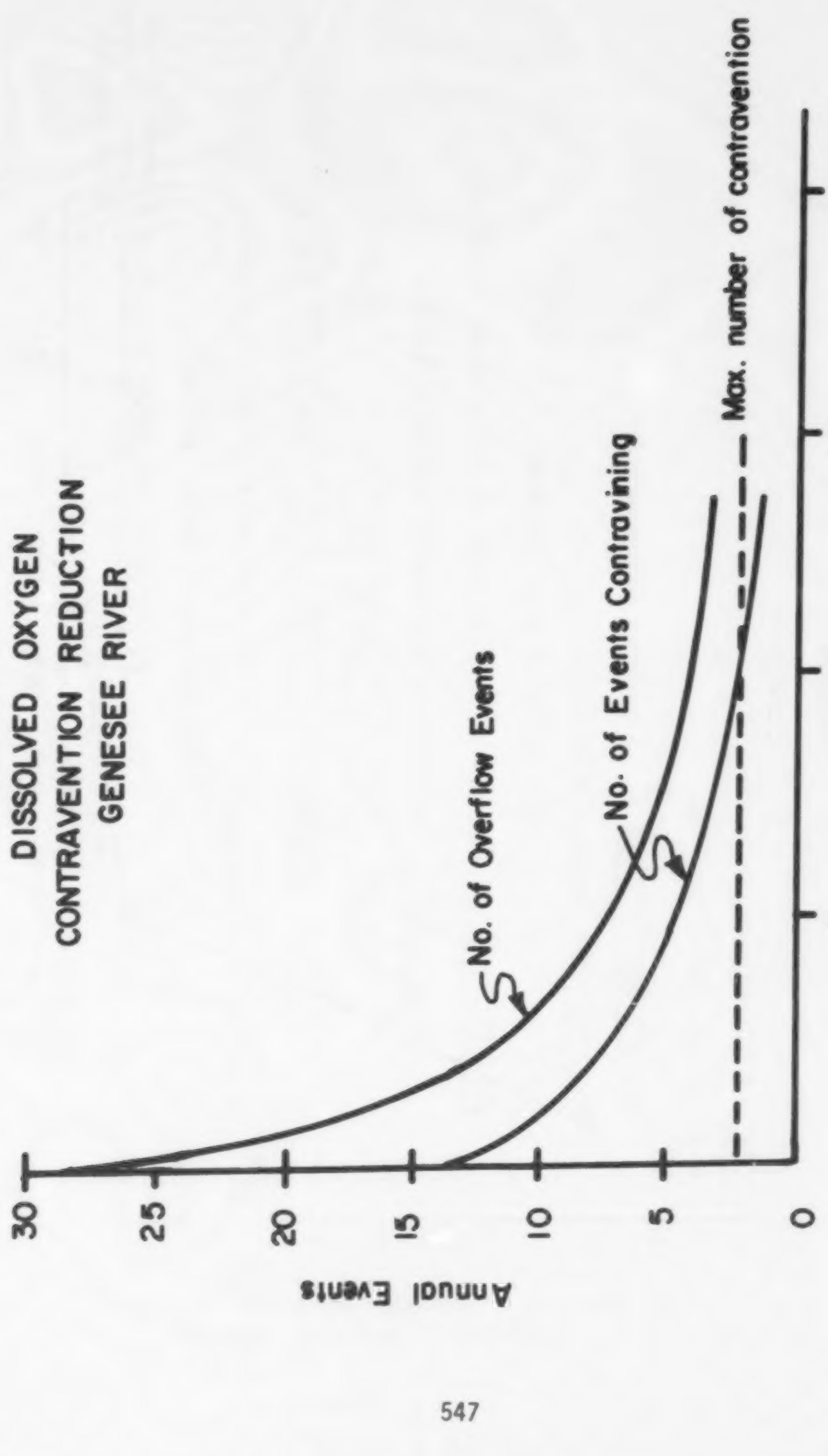


FIGURE 12. Genesee River Reduction In Potential Dissolved Oxygen Contravention Response To Implementation Of Abatement Measures

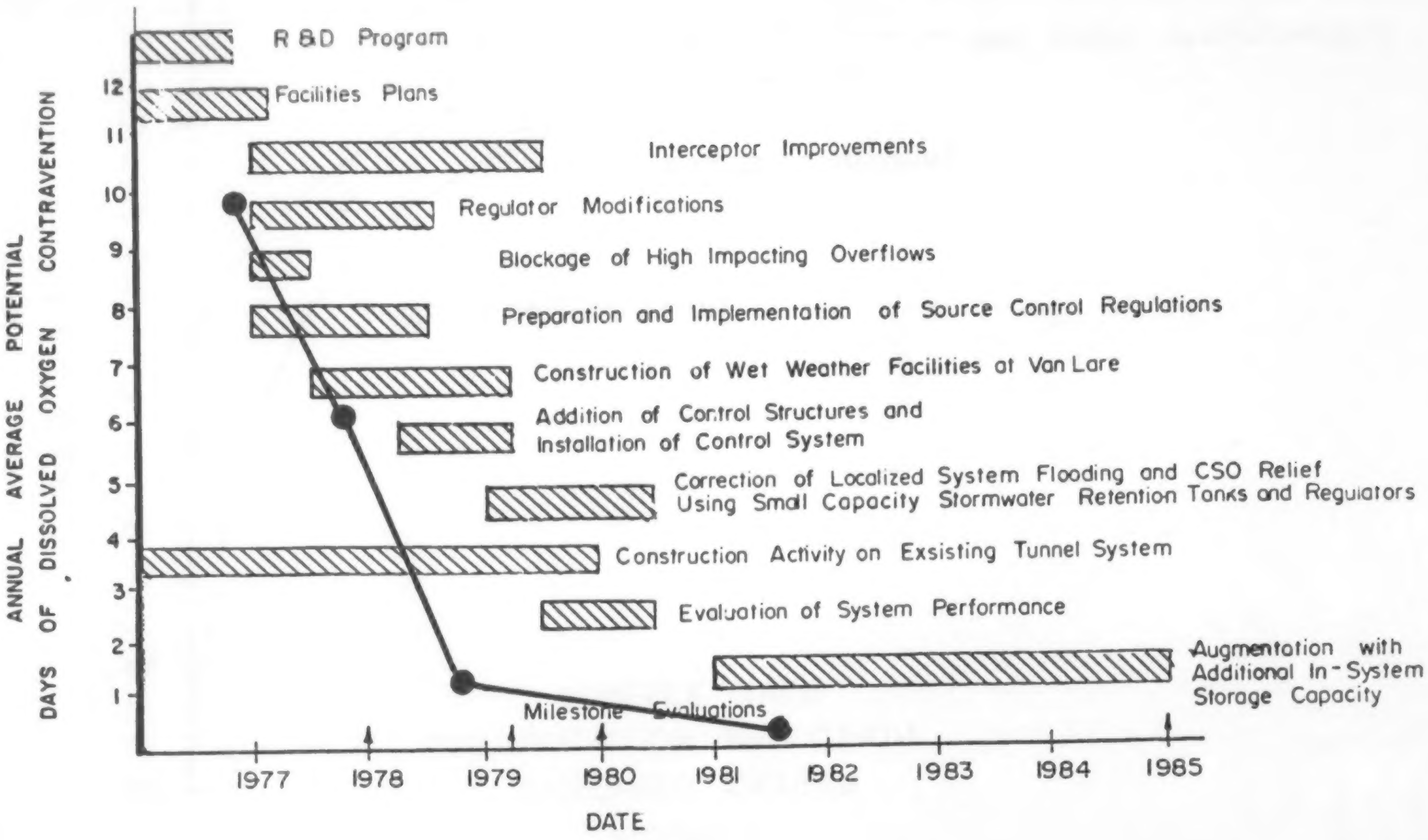


FIGURE 13. Reduction In Potential Ontario Beach Closing Days Response To Implementation Of Abatement Measures

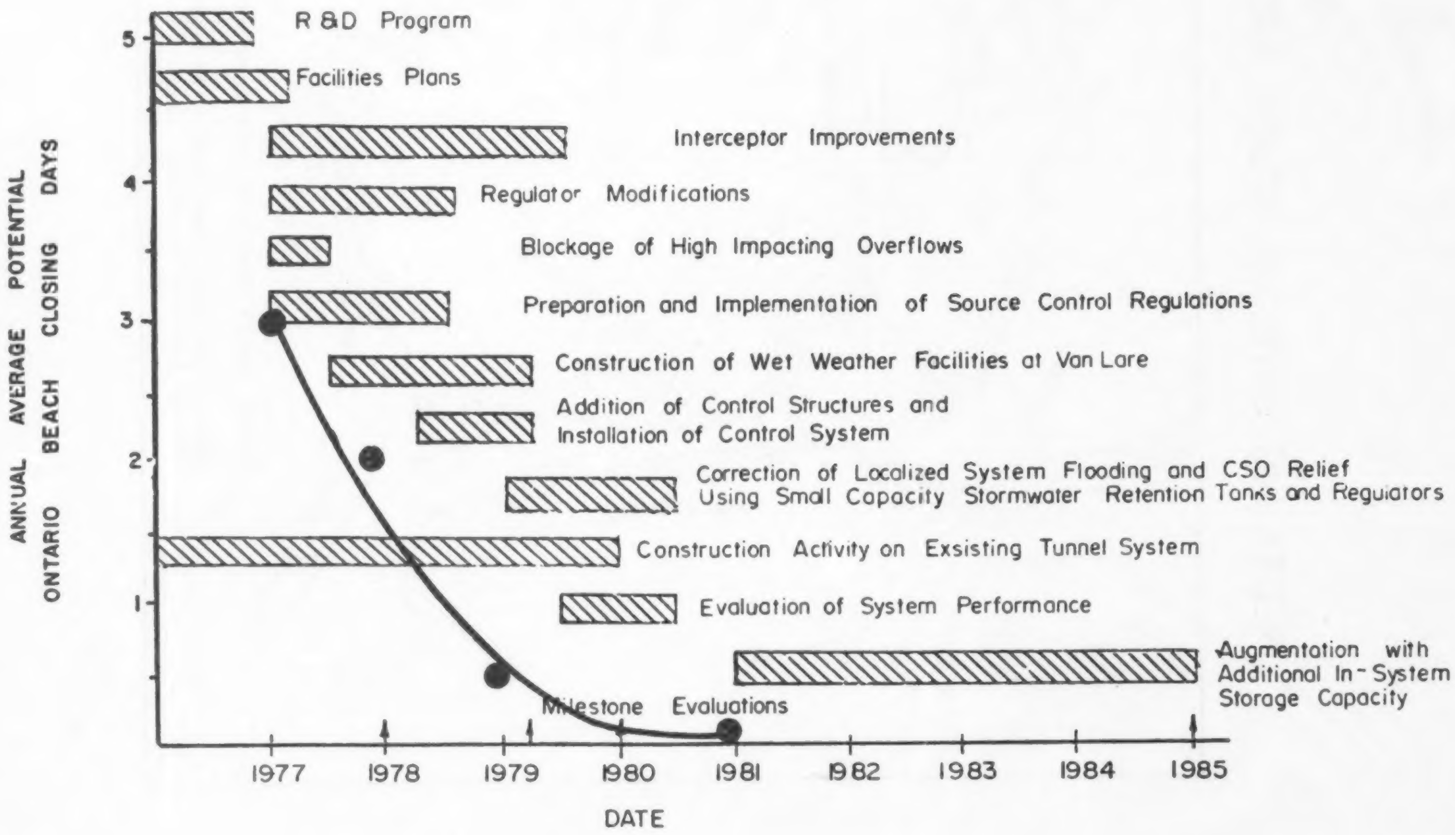
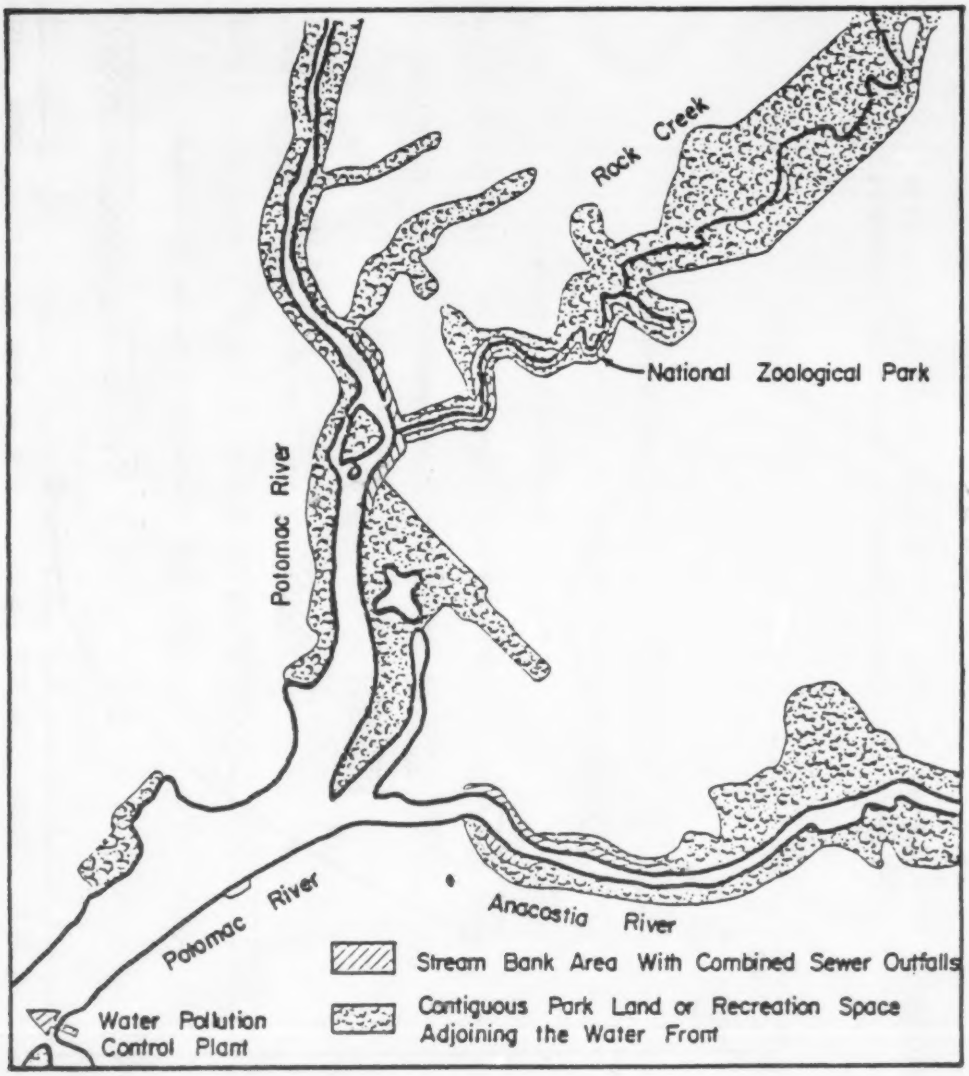


FIGURE 14

RECEIVING STREAM STUDY AREA


Rock Creek
National Zoological Park
Potomac River
Potomac River
Anacostia River
Stream Bank Area With Combined Sewer Outfalls
Contiguous Park Land or Recreation Space Adjoining the Water Front
Water Pollution Control Plant

A preliminary and simplified water quality model was used to confirm the observed water quality impacts of combined sewer overflows and establish the potential significance of other levels of combined sewer overflow loading. Model calculations indicate the combined sewer overflows can cause transient DO deficits of 3 to 5 mg/l under a variety of loading and environmental conditions. A simple Lagrangian framework, finite difference model of CBOD, NBOD, and dissolved oxygen was used by Limno-Tech in this analysis (5). The model inputs for physical data and kinetic expressions were determined from analysis by the Annapolis Field Office of EPA (6,7).

Water Quality Impacts: Potomac and Anacostia Rivers

Analysis of historical data, reconnaissance data, and preliminary model calculations were used to relate water quality problems to combined sewer overflows. Water quality problems in the Potomac and Anacostia Rivers are similar and many can be partially attributed to combined sewer overflows. Historical monitoring data for the Potomac and Anacostia Rivers revealed significant decreases in dissolved oxygen and increases in fecal coliform concentrations subsequent to rainfall events. Algal concentrations in the Potomac River were high and growth is potentially being locally stimulated by nutrients from combined sewer overflows. Estuary sediments were moderately polluted with respect to sediment oxygen demand and selected heavy metals concentrations.

Eight of twelve storm related data sets identified from the Potomac River monitoring revealed significant reductions in dissolved oxygen concentration subsequent to rainfall events, while seven of nine revealed increases in fecal coliform concentrations. In the Anacostia River five of seven storm related data sets showed decreases in dissolved oxygen and six of seven showed increases in fecal coliform. The existence of these impacts in water quality was further supported with reconnaissance data and preliminary model calculations. Overall, combined sewer overflows were shown to potentially decrease Potomac and Anacostia River dissolved oxygen concentrations by as much as 3 to 5 mg/l and increase fecal coliform concentrations in excess of 200,000 organisms/100 ml. Water quality in the Anacostia River was also persistently poor during dry weather, even though the river receives no major continuous wastewater discharges. Anacostia River water quality was also observed to be impacted by storm runoff in areas above the discharge of District combined sewer overflows.

Routine monitoring data showed high and variable concentrations of chlorophyll a in the Potomac River, near Washington, D.C. often ranging from 50 to 100 μg/l while concentrations exceeded 200 g/l in areas downstream. A comparison of the relative loadings and preliminary model calculations showed that combined sewer overflow nutrients could cause significant localized increases in Potomac River algal concentrations. Combined sewer overflows from a once in two year storm during the summer can increase nitrogen loading in the upper Potomac River by approximately 30% and phosphorus loading by 150%. On a year averaged basis the combined sewer overflow contribution was estimated to be approximately 1% for nitrogen and approximately 10 to 15% for phosphorus. Preliminary model calculations indicate that combined sewer overflow loading can significantly increase local river nutrient concentrations and that these increases can potentially increase algal concentrations beyond 100 mg/l chlorophyll-a.

Measurements of sediment conditions in the Potomac and Anacostia Rivers revealed moderately polluted conditions which may be partially caused by combined sewer overflows. Measured sediment oxygen demands averaged 2.4 g/m2/day and were elevated over values measured in the Potomac River upstream of the combined sewer overflows. Measurements of selected heavy metal conconcentrations in sediments indicated moderately to heavily polluted conditions according to EPA sediment pollution guidelines. Concentrations of lead and zinc were particularly high. No significant pollution was observed with respect to sediment concentration of oil and grease, and chlorinated pesticides.

Water Quality Impacts: Rock Creek

Analysis of historical and reconnaissance data from Rock Creek show a significant relationship between storm events and increased fecal coliform concentrations and a possible relationship to decreases in dissolved oxygen. Six of eight storm related data sets showed dramatic increases in fecal coliform subsequent to storm events but only six of eleven data sets revealed decreases in dissolved oxygen. Significant storm-related water quality impacts were also observed in Rock Creek upstream of the District combined sewer overflows.

Rock Creek sediments were measured to be moderately polluted although the extent of these sediments was limited. Again, as in the Potomac and Anacostia Rivers, lead and zinc concentrations were measured at polluted levels. Sediment concentrations of oil and grease and chlorinated pesticides were not in the polluted range. A characterization of the Rock Creek benthic macroinvertebrate community indicates a predominance of pollution tolerant organisms. No quantitative relationship could be made in the preliminary study between sediment pollution and combined sewer overflows, although such a relationship may exist. Concentrations of some heavy metals were observed to approximately double downstream of District combined sewer overflows.

Preliminary Summary of Impacts

Figure 15 presents a summary of water quality issues as they relate to the discharge of combined sewer overflows to the Potomac River, Anacostia River and Rock Creek. Analysis of historical and recent water quality data as well as preliminary calculations indicate the District combined sewer overflows significantly impact dissolved oxygen and fecal coliform in the Potomac and Anacostia Rivers and potentially impact algal and sediment concentrations. District combined sewer overflows also have potentially significant impacts on water quality in Rock Creek, although it is difficult at this time to differentiate these impacts from other upstream storm related impacts.

FIGURE 15

SUMMARY OF WATER QUALITY ISSUES

	Dissolved Oxygen	Fecal Coliform	Algal Nutrients	Sediment Chemistry
POTOMAC RIVER	●	●	◒	◒
ANACOSTIA RIVER	●	●	◒	◒
ROCK CREEK	◒	●	○	◒

LEGEND

● Major Significance - Strong Evidence of a Significant Impact From Combined Sewer Overflows

◒ Some Significance - More Limited Evidence of Significant Impact From Combined Sewer Overflows

○ No Significance - No Evidence of a Problem Area Significantly Impacted by the Combined Sewer Overflows

It has been recommended that a program of wet weather water quality monitoring be conducted on all three receiving waters along with those special field studies which will be used to develop mathematical water quality models. The Phase II program involves the development of Rock Creek fecal coliform models as well as the adaptation of the available Dynamic Estuary Model to predict the fecal coliform, dissolved oxygen and nutrient levels within the Anacostia and Potomac Rivers. These models will provide a quantification of the incremental improvements in water quality resulting from various combined sewer overflow control alternatives.

Methodology of Water Quality Utilization

Now that the application of water quality models has been presented for the Rochester, New York and Washington, D.C. combined sewer overflow planning studies, it would be useful to summarize a methodology which has evolved from conducting these studies. Figure 16 presents a summary of 17 steps which constitute the process leading from the initial review and compilation of the available water quality data base to the marginal benefit/marginal cost analysis and sizing of the most cost-effective solution.

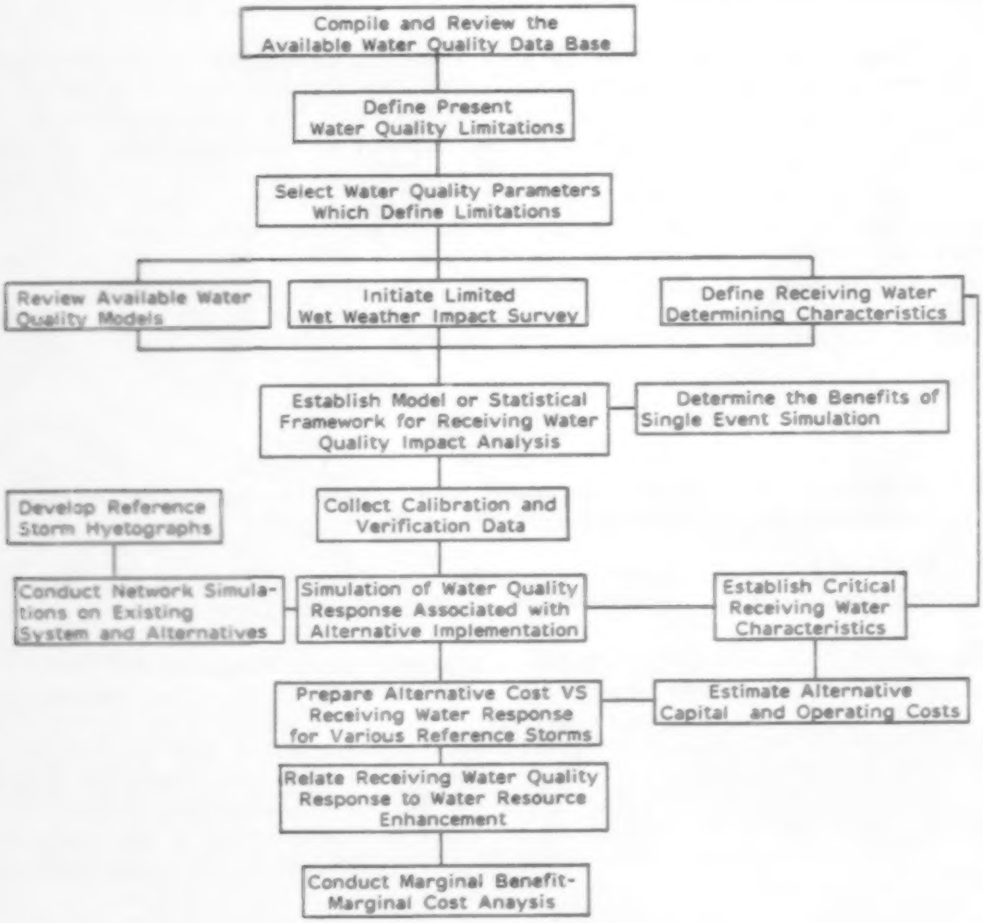
In the initial phase of the analysis, considerable effort is required in the definition of the baseline water quality limitations and the selection of appropriate water quality parameters which can be utilized to define the established limitations. Based on this initial definition a limited wet-weather impact survey should be conducted to confirm and further define the urban runoff induced water quality limitations. In conjunction with this limited wet-weather impact survey the available water quality models should be reviewed and impact-determining receiving water characteristics (low flow, tidal influence, unique sources and sinks, etc.) defined.

After the initial definition of the system, decisions relative to the impact analysis framework should be resolved. If sufficient data are available, a regression relationship may be developed which relates a water quality parameter to several determining variables. As an alternative a deterministic rather stochastic model may be required. A limited data base may require the acquisition of water quality impact data; the specific parameters and the extent of the data base is largely determined by the complexity of the receiving water system and type of modeling (steady state, dynamic, etc.). The collected data are required for calibration and verification of the selected modeling technique.

Following selection, calibration, and verification of the water quality analysis framework, simulation of both present and improved water quality conditions may be conducted given the urban runoff loadings projected for existing and improved conveyance and treatment systems. A method of simulating the stormwater and/or combined sewer loadings is required. Available conveyance network simulators include Storm, the Simplified Stormwater Model (SSM) and the various versions of the EPA

FIGURE 16

METHODOLOGY OF WATER QUALITY MODEL UTILIZATION FOR CSO MARGINAL BENEFIT-MARGINAL COST ANALYSIS


Compile and Review the Available Water Quality Data Base
Define Present Water Quality Limitations
Select Water Quality Parameters Which Define Limitations
Review Available Water Quality Models
Initiate Limited Wet Weather Impact Survey
Define Receiving Water Determining Characteristics
Establish Model or Statistical Framework for Receiving Water Quality Impact Analysis
Determine the Benefits of Single Event Simulation
Develop Reference Storm Hyetographs
Collect Calibration and Verification Data
Conduct Network Simulations on Existing System and Alternatives
Simulation of Water Quality Response Associated with Alternative Implementation
Establish Critical Receiving Water Characteristics
Prepare Alternative Cost VS Receiving Water Response for Various Reference Storms
Estimate Alternative Capital and Operating Costs
Relate Receiving Water Quality Response to Water Resource Enhancement
Conduct Marginal Benefit-Marginal Cost Anaysis

Stormwater Management Model (SWMM) which can be well adapted to specific requirements to provide a dynamic or steady state loading.

Coincident with the development of system loadings is the necessity to establish the critical or representative water quality conditions (water column temperature, receiving water flow, tidal cycle, etc.). It may be necessary to statistically analyze the probability of occurrence of defined critical receiving water conditions along with the probability of generating a rainfall event of given intensity and duration. Establishment of historical critical receiving water conditions as the basis for water quality simulations can result in a significant overstatement of both the problem and the facilities required to solve the problem.

Simulation of the water quality response associated with alternative implementation can be accomplished given the defined receiving water conditions, urban runoff generated loadings and the calibrated water quality simulator. A progressive development of improved water quality parameters along with the associated capital and operating costs allow the construction of a cost curve vs water quality improvement for various sized facilities. From this cost vs water quality index curve the marginal increase in receiving water benefit can be associated with an increase in marginal cost. Utilizing reasonable judgement, the point where marginal benefits outstrip marginal costs can be assessed within a band of accuracy established by the various simulators.

REFERENCES

1. Drehwing, F.J., et al., Combined Sewer Overflow Abatement Program, Rochester, N.Y., Volume I, EPA Grant No. Y005141, 1979.

2. O'Brien & Gere Engineers, Inc., Genesee River Water Quality Investigations, Part I, Joint Venture, Rochester, New York, 1976.

3. Canale, R.P., and Nachiappan, S., Steady State Modeling Program, Sea Grant Technical Report N. 27, University of Michigan, March, 1972.

4. Bonham-Carter, G., Thomas, J.H. and Lockner, D., A Numerical Model of Steady Wind-driven Currents in Lake Ontario and the Rochester Embayment Based on Shallow Lake Theory, IFYGL Rochester Embayment Project Report #1, Department of Geological Sciences and Department of Mechanical and Aerospace Sciences, University of Rochester, Rochester, New York, 1973.

METHODOLOGY FOR EVALUATING THE IMPACT AND ABATEMENT OF COMBINED SEWER OVERFLOWS
A CASE STUDY OF ONONDAGA LAKE, NEW YORK

and

Peter E. Moffa
John C. Byron
Steven D. Freedman
Stearns & Wheler, Civil and Sanitary Engineers
Cazenovia, New York 13035

and

John M. Karanik
Randy Ott
Onondaga County
Department of Drainage and Sanitation
North Syracuse, New York 13212

ABSTRACT

A general methodology is presented for the evaluation of the impact and abatement of combined sewer overflows on receiving waters. It was developed from experience with Onondaga Lake in Central New York which receives combined sewer overflows from the City of Syracuse.

Field investigations of the combined sewer system and the receiving water must first be undertaken. The field work includes flow measurement and water quality sampling of the sewer overflows and the receiving water during several different storms. Use of a computerized data bank has been found virtually essential for the storage and manipulation of the large quantity of data resulting from the sampling and analysis.

Mathematical modeling of the receiving water is undertaken to evaluate water quality as a function of pollutant load; the storm sewer system is modeled to determine the quantities of pollutants discharged during various storm conditions.

Abatement alternatives, and their respective costs, for the reduction of pollutants from wet-weather sources, particularly combined sewer overflows, are next investigated. Using engineering judgment for the most effective and economic abatement measures, a relationship is then developed between abatement cost and storm condition for each of several water quality criteria or goals. From the cost-benefit relationships thus developed, a graphical determination can be made of the "general optimum solution" (GOS) for reduction or treatment of combined sewer overflows.

Abatement alternatives and their respective costs are investigated next for the reduction of pollutants from wet-weather sources, particularly combined sewer overflows.

In the study for Onondaga County, New York, 35 overflows from the combined sewers of the City of Syracuse, which serve an area of about eight square miles, were monitored for a period of one year. Onondaga Lake is approximately four and one-half square miles in surface area; it was sampled at ten surface locations for the period of influence of each of six storms. The U. S. Environmental Protection Agency Storm Water Management Model (SWMM) was applied to the City's sewer system. A 27-segment, three-dimensional, dynamic water quality model of the lake was developed with capability of predicting enteric bacteria, dissolved oxygen, nutrients, and toxic materials.

From the models, it was determined that the impact of combined sewer overflows (CSO) on dissolved oxygen concentrations in Onondaga Lake will not be critical after tertiary treatment facilities for dry-weather wastewaters are placed in operation. Combined sewer overflow contributions of phosphorus will be negligible in comparison to those from other sources.

In an average rainfall year, 38 violations of the fecal coliform standard will occur in the area of the lake intended for contact recreation. If abatement of CSO pollution were to follow the "general optimum solution" of this methodology, there would still be 13 annual violations, ten of which would occur from June through September. Inasmuch as each violation persists for about three days, more extensive CSO abatement will be required if the projected recreational usage of Onondaga Lake is to be realized.

The methodology was prepared in fulfillment of (U. S. Environmental Protection Agency) Contract No. R805096010 by Stearns & Wheler, Civil and Sanitary Engineers, and the Onondaga County Department of Drainage and Sanitation.

INTRODUCTION

The impetus provided by the Federal laws concerning water pollution (1, 2) has resulted in great strides in wastewater treatment. The prevailing attitude leading to the 1972 Amendments to the Water Pollution Control Act (1) was that there had been a general lack of progress in abating the country's pollution. Consequently, secondary treatment plants throughout the country were built on the basis of the obvious benefit they would provide to receiving water quality.

Since 1972, the required "201" facilities planning has progressively expanded to include analyses and evaluations of separate sewer systems, and most recently in some states, combined sewer systems. The combined sewer overflow (CSO) investigations have had to address an old but relatively undefined component of municipal wastewater; namely, stormwater. The highly variable nature of this element and its double-edged impact on receiving water of providing dilution as well as adding pollutants has resulted in a more deliberate approach to cost-benefit relationships than had previously been the case for facilities planning.

The common "dry-weather" receiving water standard based on consecutive seven-day low flow conditions occurring once every ten years (MA7CD10) for conventional treatment has gradually given way to the consideration of allowing deviation from "dry-weather" for certain storm events. Recently, attention has been devoted to the concept of the "design storm" as being that storm for which abatement facilities are planned (3). However, the elusiveness of this approach has led to the consideration of cumulative storm effects over monthly or annual cycles. A defined approach is not yet available, but actual ongoing experiences are now providing the basis for practical approaches to the problem.

The methodology outlined herein is based largely on, but not limited to, the successful experience in identifying the discharges from a combined sewer system and their impact on the water quality of an urban lake in Onondaga County just north of Syracuse, New York (4).

The general tasks involved in a CSO study are shown in Figure 1. The level of effort of each task will vary depending upon (1) the type of

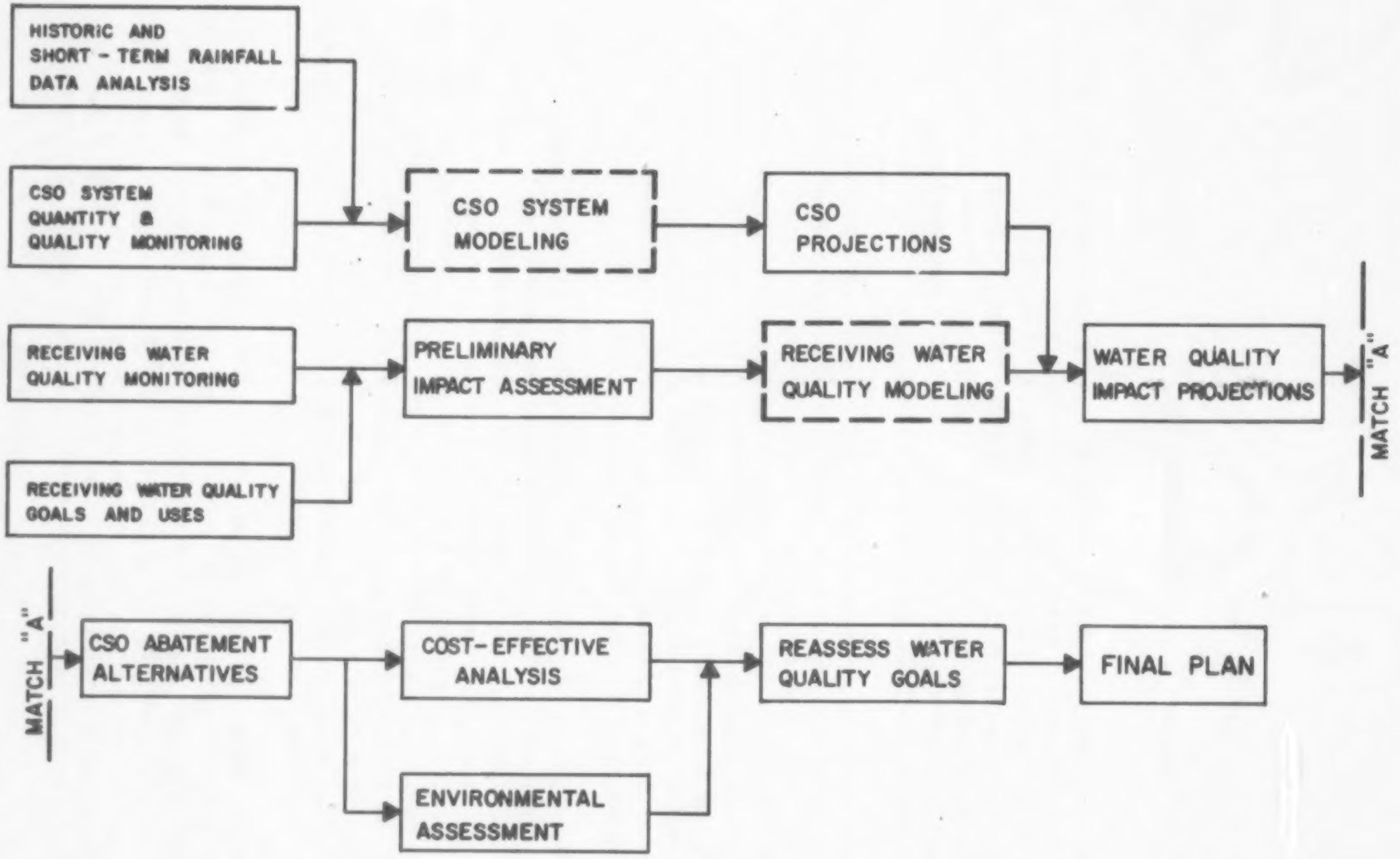

HISTORIC AND SHORT - TERM RAINFALL DATA ANALYSIS
CSO SYSTEM QUANTITY & QUALITY MONITORING
RECEIVING WATER QUALITY MONITORING
RECEIVING WATER QUALITY GOALS AND USES
CSO SYSTEM MODELING
PRELIMINARY IMPACT ASSESSMENT
CSO PROJECTIONS
RECEIVING WATER QUALITY MODELING
WATER QUALITY IMPACT PROJECTIONS
MATCH "A"
MATCH "A"
CSO ABATEMENT ALTERNATIVES
COST-EFFECTIVE ANALYSIS
ENVIRONMENTAL ASSESSMENT
REASSESS WATER QUALITY GOALS
FINAL PLAN

Figure 1. Essential tasks of CSO abatement studies.

assessment required; (2) the uses of the receiving waters; (3) the budgetary limitations of the study; and (4) other factors particular to the study area.

COMBINED SEWER OVERFLOW MONITORING

The objectives of any wet-weather monitoring program are not to capture the "elusive" critical storm conditions, but to sample a statistically significant number of storms to be able to develop relationships between rainfall-runoff-overflow and receiving water quality for the ultimate purpose of projecting cause-effect relationships. Short-term rainfall events provide the "real time" data for impact assessment while long-term or historic rainfall data provide the statistical basis for making projections.

CSO system monitoring and data gathering can be greatly assisted by (1) interviews with knowledgeable municipal personnel; (2) previous reports on the system, such as Infiltration/Inflow Analyses and Sewer System Evaluation Surveys; and (3) record drawings on the interceptors and diversion structures.

The measurement of overflow quantity may involve the measurement of the flow reaching the interceptor as well as the flow discharged to the receiving water, depending upon the capacity of the interceptor. The number of individual points that have to be measured should be minimized to reduce equipment maintenance and data coordination. It also is desirable to install the flow metering device at the same location as the automatic sampler which measures quality so that operation of the sampler will be triggered by an actual overflow occurrence. Such time coordination becomes critical for determining pollutant loads, which are calculated as the product of flow quantity and pollutant concentration. Thus, the simplest and most direct measurement is at the overflow to a receiving water. Under surcharge conditions, it is particularly necessary to estimate the interceptor flow as well as the receiving water overflow in order to determine the true quantity of overflow originating from a tributary area.

The frequency and duration of sampling should be based on an adequate description of the peak period of pollutant discharge, often called the first-flush period, and the subsequent subsidence of this period to relatively insignificant quantities. It has been found that in the period of five minutes following the onset of runoff, significant concentrations of pollutants can occur. Thereafter, 7- to 15-minute intervals have proven to be adequate to define an overflow discharge. In general, the minimum duration of sampling should be ample to capture the runoff from a six-hour storm.

Table 1 shows some of the more important factors affecting CSO quality.

ONONDAGA COUNTY

The combined sewer system of the City of Syracuse covers approximately eight square miles and has 87 active overflow sites, of which 35 were monitored for flow quantity and pollutant quality. The monitored sites encompassed 85 percent of the total defined runoff area and hence were representative of the overall system. Six rain gages were located throughout the City to monitor rainfall, each representing an area of approximately two square miles.

The main interceptor system has little or no capacity for wet-weather flow and thus the quantity of flow discharged to the stream was assumed to be the total quantity entering the diversion structure. This assumption may not be applicable in many cases and must be verified by field data.

Overflows were monitored through the use of ultrasonic flow meters in combination with sampling units capable of taking 24 separate samples. Samplers were initiated by a signal from the flow meter and were operated at 7-1/2 minute intervals thereafter (4).

COMBINED SEWER OVERFLOW MODELING

CSO system modeling is also an important task, and one which can be troublesome if not approached cautiously. As the complexity of the urban stormwater model increases, so does (1) the data needs for calibration and verification; (2) the associated costs for obtaining the data; and (3) the cost of operating the mathematical model. A good rule to follow in selecting stormwater models is to use the simplest one which will produce the desired results. The major types of data which are required for most models include (1) surface area and slope; (2) percent impervious; (3) major collection system components, such as sizes and slopes; (4) overflow hydraulics; and (5) rainfall.

There are a number of mathematical models currently available for CSO and stormwater analyses. They range from the "desk-top", hand-calculated models, such as the simple "rational formula", to the sophisticated EPA Storm Water Management Model (SWMM) (6, 7). Three categories have been suggested for all stormwater assessment models, based on their applications, as shown in Table 2.

TABLE 1. FACTORS AFFECTING CSO QUALITY

Antecedent Dry Weather	Affects amount of pollutants available on surfaces and in sewers, and sets initial infiltration capacities of previous surfaces.
Land Use of Catchment Area	The type of land use can dramatically affect the pollutant accumulation rates in an area.
Pollutant Accumulation and Washoff	Accumulation is dependent on antecedent dry weather, land use, street sweeping practices and rainfall characteristics. Pollutants are often modeled as fractions of the amount of dust and dirt.
Street Sweeping Practices	Affects the amount of solids on the streets which is available for washoff
Soil Erodibility and Erosion Control Practices	Erosion is a source of solids in combined sewers.
Dry Weather Wastewater	Determines the background quantity and quality within an area.
Solids Deposition and Scour in Sewer System	Dependent on antecedent dry weather, sewer system characteristics and rainfall characteristics.
Sewer System Characteristics	Slope, shape and size affect solids transport and deposition.

TABLE 2. CATEGORIES OF STORMWATER MODELS

Level	Application
I	Problem Assessment
II	Planning
III	Event Analysis

Table 3 further describes the three categories of stormwater assessment in terms of model complexity and characteristics.

Some models are capable of being used for more than one application. For example, the EPA SWMM model is considered a single event, or Level III model, but it can also be operated continuously, or as a Level II model. Those models which do not require the use of computers, such as the EPA-Level I SWMM, the URS, Inc. procedure, and the EPA Areawide Assessment Procedures Manual method, are considered Level I, or problem assessment models (8, 9, 10, 11). In addition to the continuously operated EPA SWMM, other planning, or Level II, models include the U. S. Army Corps of Engineers' STORM and EPA Simplified SWMM (SSWMM) (11, 12).

RECEIVING WATER MONITORING

Receiving water monitoring is one of the most important aspects of the data gathering efforts. While data may exist on both dry-weather conditions and point source discharges, there rarely is available adequate wet-weather data. Dry-weather data, although providing an important comparative basis, rarely describes wet-weather responses.

A multitude of pollutants can be introduced into a receiving water from CSO's, however, the primary concerns can be categorized into (a) pathogens, because of public health, and (b) dissolved oxygen, nutrients and toxics, all because of their direct effect on the aquatic system and indirect effect on public health. The fecal coliform and fecal streptococcus groups can be used as pathogenic indicators. Dissolved oxygen, phosphorus, and, in some cases, nitrogen, are critical nutrients or indices to be evaluated for their impact on aquatic life. Initially, evaluation of toxics should be limited to the heavy metals, unless specific information indicates the need to investigate toxicity from organic chemicals, e.g., pesticides, PCB's. Organic analyses represent a relatively high level of complexity. Toxicity observed in the BOD test can be used as an indicator for proceeding with any additional analyses. One exception to this dictum would be the case where bottom deposits that may be subject to toxic organic

TABLE 3. STORMWATER ANALYSIS LEVEL AND MODEL COMPLEXITY

Analysis Level	Model Type	Model Complexity	Purpose of Model	Model Characteristics
I	Desk-top	Low to medium	Problem assessment, preliminary planning, alternative screening	No computers. Equations, nomographs based on statistical analyses of many years of records.
II	Continuous simulation	Low to medium	Problem assessment, planning, preliminary sizing of facilities (particularly storage), alternative screening. Assess long-term impacts of designs.	Program of few hundred to few thousand statements. Uses many years of rainfall records with daily time steps, or worst two years with hourly time steps. May include flow routing and continuous receiving water analysis.
III	Single event simulation	Medium to high	Analysis for design, detailed planning.	Program to cover 10,000 statements. Higher modeling precision, from rainfall through sewers, possibly to receiving waters. Short-time steps and simulation times. Fewer alternatives to be evaluated.

contaminants are scoured frequently and play a significant role in the year-round quality of overlying waters.

Decisions as to what storms should trigger receiving water sampling should be based on the anticipated storm intensity. Close coordination should be maintained with those sampling the sewer system so as to maximize the number of overflows being monitored during receiving water sampling. Sampling locations and the frequency and duration of sampling are dependent upon the changes a particular parameter may undergo during a storm period. Thus, prior knowledge from other studies or background data on the receiving water is helpful. In the absence of such information, the general guidelines outlined herein can be used.

The location of sampling points should reflect the intended usage within the body of the receiving water. In general, areas where contact recreation is anticipated should be sampled more intensely than areas where fishing or boating are anticipated. The circulation pattern of the receiving water should be considered in determining surface locations and depths of sampling. Surface locations should include tributaries as well as in-stream locations reflecting the outer boundary of any such tributary's mixing zone. In this way, the impact of a particular tributary can be estimated. If a tributary receives combined sewer overflows, it should be sampled so as to bracket these discharges, either individually or collectively.

ONONDAGA COUNTY

Water quality data from dry-weather periods were available for Onondaga Lake and were used to estimate the pollutant levels expected during storms. In actuality, these estimated levels were often exceeded.

Anticipated storm intensity determined whether or not receiving water sampling was to be conducted for any particular storm. On each positive occasion, the crews in charge of sampling the sewer system were alerted. The frequency and duration of sampling during a storm were estimated at the onset of the project, but were later adjusted on the basis of the first two storms. The water quality parameters generally measured were bacteria, BOD, DO, solids, nutrients, and heavy metals.

All major tributaries were sampled and measured at their points of entrance into Onondaga Lake. In addition, two of the tributaries were sampled upstream of the CSO's so as to bracket its collective input. Preliminary sampling was performed to determine the validity of sampling techniques for each of the tributary locations. To be representative, tributaries were sampled in cross section and in depth, depending upon their dimensions and the degree of mixing in each. The lake was sampled at ten surface stations; each station was sampled at two depths to reflect the epilimnetic (upper) and hypolimnetic (lower) waters. One surface station was located at each of the lake's two deepest points, the remainder being located to reflect tributary inputs or shoreline areas expected to be used for contact recreation. Figure 2 illustrates the sampling locations.

DATA BANK

Whenever CSO studies yield large amounts of data, it is important that they be handled in a form that is easily retrieved and analyzed. For any kind of system, every sample must be adequately identified, e.g., storm number, location, date, time, and parameter. A data file may take the form of summary sheets or can be computerized, depending upon the size of the project. A computerized file will permit the rapid retrieval of data in various forms, which is particularly helpful in CSO projects where dilution volumes of samples for certain parameters may have to be adjusted following the first few storm samplings. These and other adjustments, such as monitoring locations and possible elimination of certain stations, require that initial data be evaluated as soon as possible. Provision should be made to plot the concentration levels of the various parameters versus time for a given CSO and, when combined with flow, calculate a plot of pounds of material versus time. Other sub-routines could be developed to determine the pounds of material discharged during an entire storm.

ONONDAGA COUNTY

A computerized file system was established for the combined sewer system and a separate file was set up for the receiving water. Since the combined sewer system investigation and the receiving water investigations were conducted by independent organizations, separate data systems were developed; this presented no problem since, with the exception of storm sampling, the data analyses were relatively independent of each other. The output requirements and mathematical modeling for the sewer system were markedly different from those of the receiving water.

RECEIVING WATER MODELING

Receiving water modeling must be tailored to the particular situation. For example, simple steady-state biochemical oxygen demand-dissolved oxygen (BOD-DO) models should first be used to assess impact rather than more complex dynamic models. However, in the long run more detailed modeling may be required. Simple fecal coliform die-off models, such as Chick's Law, can be used effectively to make preliminary assessments. When an estuary or lake is involved, the receiving water

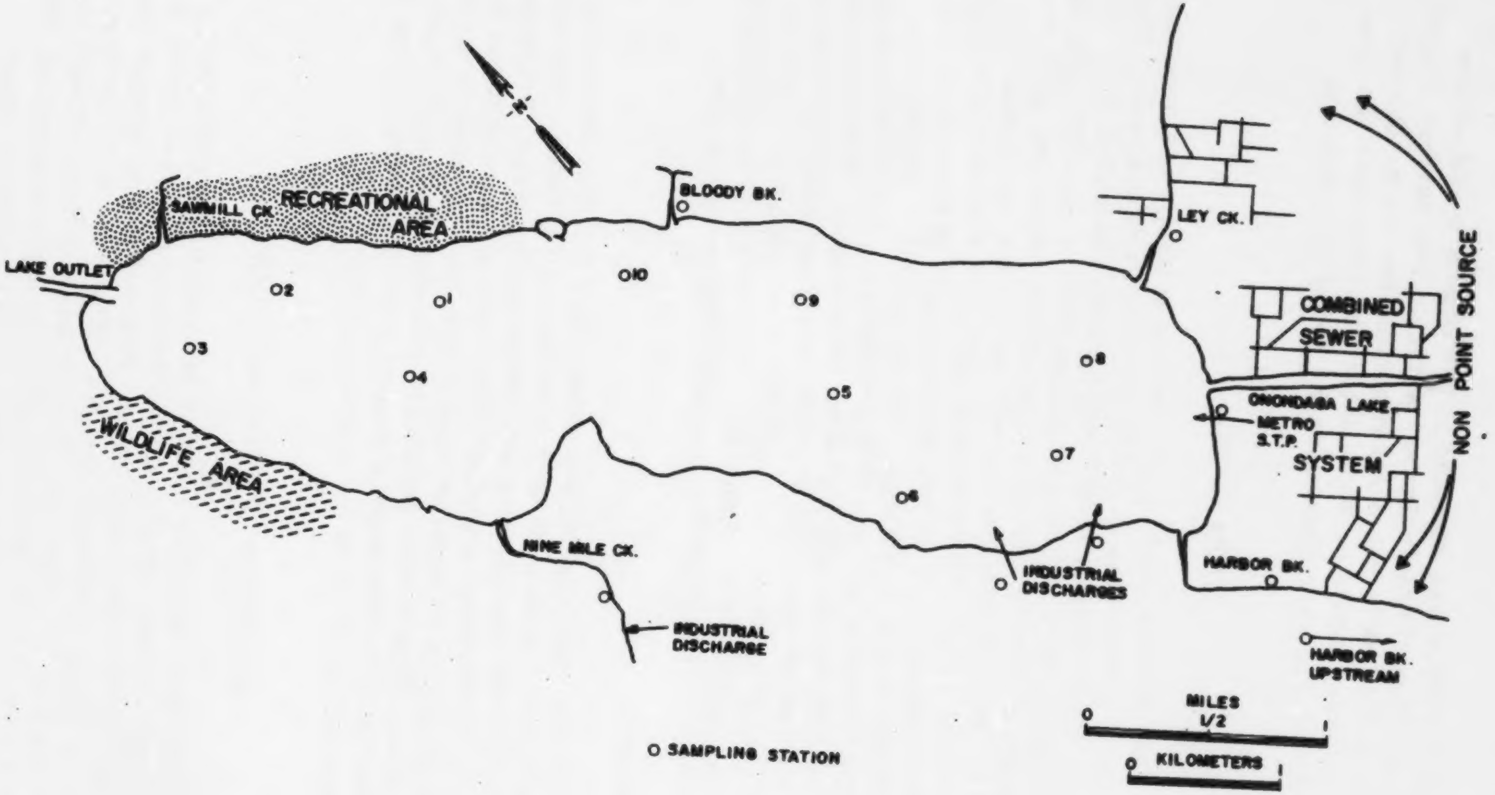


Figure 2. General plan and sampling locations for Onondaga Lake. (18)

assessment may be slightly more complex than a flowing water body, but the same "simple model first" rule should be followed.

A receiving water model should include certain fundamental functions as well as those required to meet specific needs. The model should consider circulation as fundamental to the distribution of pollutants in the receiving water. The circulation model should include advective and dispersive transport and, in addition, mechanisms for settling and scour.

ONONDAGA COUNTY

Both simplified and detailed modeling was performed for Onondaga Lake. Simplified modeling of a one-year storm showed contravention of bacterial standards throughout the recreational zones of Onondaga Lake. In order to verify these results, more detailed modeling had to be undertaken.

Simplified modeling was also attempted to determine the dissolved oxygen relationships in the lake. However, owing to the complexity and intensity of algal growths and their significant influence on the DO level, a more detailed model was required.

On the other hand, a simplified approach was taken to determine the significance of phosphorus loadings from storm runoff on an annual basis. This approximation was considered adequate to assess what turned out to be the relatively insignificant contribution of stormwater to the phosphorus content of the lake.

Each CSO was identified as entering one of two tributaries to the lake. In the detailed receiving water model, the CSO's entering each tributary were treated as single, collective inputs. The lake was divided into 27 distinct segments for the purpose of defining circulation patterns. Twenty-one (21) of the segments were located in the epilimnetic (upper) waters and six of the segments were located in the hypolimnetic (bottom) waters, as illustrated by Figure 3. The model utilized steady-state circulation patterns and hydraulic loads in combination with time-variable pollutant loads to simulate the lake's water quality response to stormwater discharges.

The model described the three-dimensional transient distribution of fecal coliform, total phosphorus, BOD and dissolved oxygen. The model included advective and dispersive transport and mechanisms for settling, scour, and chemical and biochemical reaction.

The model is based on the conservation of mass as expressed by the following equation:

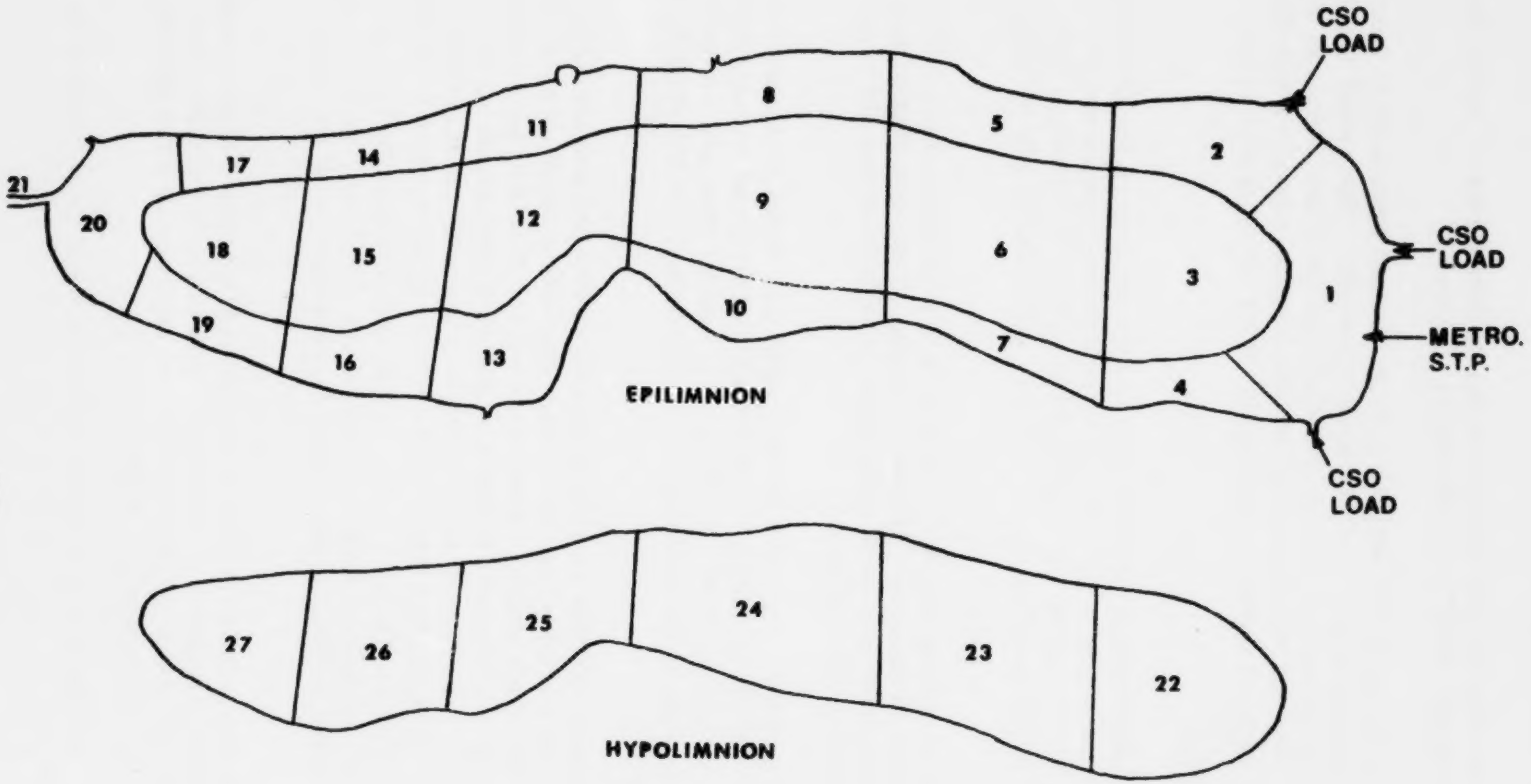


Figure 3. Model segmentation.(18)

$$\underset{\text{of Mass}}{\text{Accumulation}} = \underset{\substack{\text{from Advective}\\ \text{Transport}}}{\text{Net Input of Mass}} + \underset{\substack{\text{from Dispersive}\\ \text{Transport}}}{\text{Net Input of Mass}}$$

$$\pm \text{ Sources and Sinks of Mass}$$

or mathematically $$\frac{\partial c}{\partial t} = \nabla . E \nabla C - \nabla . UC + r$$

where C = concentration of pollutant (mg/l)

t = time (days)

∇ = del operator

E = dispersion coefficient ($1/m^2$/day)

U = velocity (m/day)

r = reaction sources or sinks (mg/l/day)

Since a direct solution of this equation is usually not possible, the solution requires the use of approximate solutions. The finite difference solution, which is equivalent to representing the continuous lake system as a collection of completely mixed cells that are interconnected by advective and dispersive transport, was used in this model. Figure 4 illustrates the essential factors. While this approach sacrificed some spacial accuracy, the careful segmentation of the water body to closely represent the overall circulation and mixing minimized these inaccuracies. The non-steady continuity equation is developed elsewhere (13).

The Onondaga Lake model was calibrated and verified against four distinct storms. The resulting model was then used to predict two of the same storms and, in both cases, resulted in good fits. In addition, sensitivity analyses were performed for the various rate coefficients. In general, 25 to 50 percent changes, plus and minus, in the coefficients originally employed were used to test the model's sensitivity to such values in terms of water quality impact.

MODEL PROJECTIONS

Prior to projecting abatement requirements, the critical environmental conditions must be determined for use in the water quality model. Input conditions must be carefully selected so as to be critical for the parameter in question. For example, the effect of fecal coliforms on water quality is most severe under conditions of minimal die-off (low tempera tures) and rapid transport; the dissolved oxygen parameter, on the other hand, is most adversely affected when biochemical rates are maximum (high temperatures) and reaeration is minimum. Bacterial die-off rates

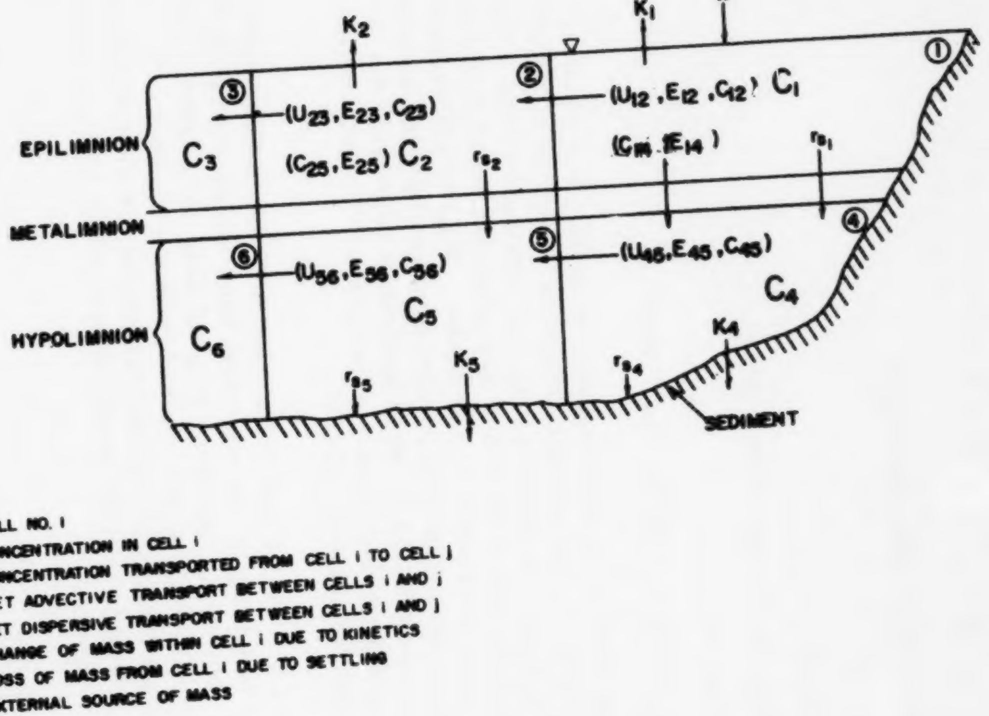

K2
K1
W
① ② ③ ④ ⑤ ⑥
EPILIMNION
METALIMNION
HYPOLIMNION
C3
C6
(U23, E23, C23)
(C25, E25) C2
(U56, E56, C56)
C5
(U12, E12, C12) C1
(C14, E14)
(U45, E45, C45)
C4
rs1
rs2
rs4
rs5
K4
K5
SEDIMENT
①-CELL NO. i
Ci - CONCENTRATION IN CELL i
Cij - CONCENTRATION TRANSPORTED FROM CELL i TO CELL j
Uij - NET ADVECTIVE TRANSPORT BETWEEN CELLS i AND j
Eij - NET DISPERSIVE TRANSPORT BETWEEN CELLS i AND j
ri - CHANGE OF MASS WITHIN CELL i DUE TO KINETICS
rsi - LOSS OF MASS FROM CELL i DUE TO SETTLING
W - EXTERNAL SOURCE OF MASS

Figure 4. Forces acting on a pollutant mass discharged to a two-dimensional advective-dispersive system

should be determined through laboratory-controlled tests on the receiving water. In predicting the dissolved oxygen content of productive lakes, light conditions and the algal standing crop can represent the dominant effect on the DO concentration.

The projection of pollutant loads requires considerable judgment to avoid a combination of events that may have a very infrequent reoccurrence interval. If, for example, an MA7CD10 dry-weather stream flow is combined with storm runoff having a recurrence interval of one year, the probability of both conditions occurring at the same time can exceed once in 30 years. The selection of dry-weather flow conditions for a large lake is less critical than for a stream because circulation is more influenced by lake elevations and lunar effects than incoming tributaries. In addition to flow considerations, pollutant loadings in a model must include dry-weather components representing both point and background sources. Point source information often can be obtained through the NPDES or State permit system, but background source information may require field measurements during dry periods.

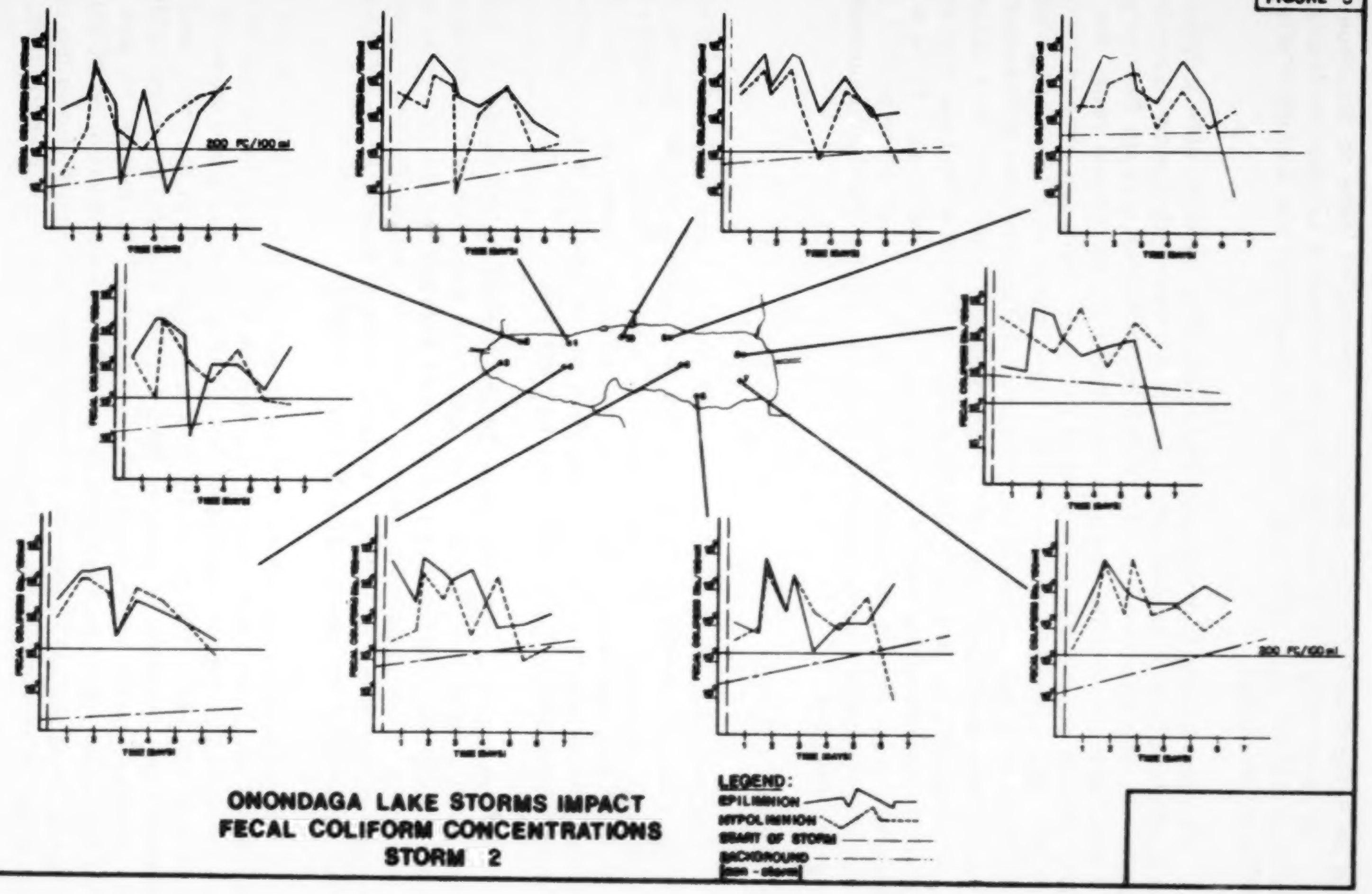
ONONDAGA COUNTY

Current measurements taken on Onondaga Lake during the storm sampling periods served as the basis for determining the in-lake circulation patterns assumed in the mathematical model. Background water quality data was available from prior dry-weather sampling. Light conditions were determined from regional data in conjunction with percent-cloud-cover measurements at the local weather station.

Dry-weather data from prior years served as the basis for determining point source pollutants. The data used were limited to the months of June through September and represented average values of two to six years of data, depending upon whether or not the data reflected contemporary operations in the tributary. In some cases, modification of the sewer system prevented the use of all six years of data.

Projections of the water quality impact of storms with recurrence intervals of one, two, and ten years were made. It was found that the one-year, two-hour storm would result in bacterial numbers in excess of the standard of 200 cells/100 ml in all "upper" segments of the lake, and a maximum concentration of greater than 11,000 fecal coliform cells/100 ml in the northern basin where contact recreation is anticipated. It was predicted that a storm of this magnitude would cause contravention of coliform standards for a period in excess of three days. Figure 5 shows the results of an actual two-year storm in terms of fecal coliform concentration.

FIGURE 5



ONONDAGA LAKE STORMS IMPACT
FECAL COLIFORM CONCENTRATIONS
STORM 2

On the other hand, the dissolved oxygen demand from even a ten-year two-hour storm had only minimal effect on the dissolved oxygen balance of the lake; the maximum DO deficit for that storm, under the most critical conditions, was determined to be 2.8 mg/l. This is shown in Figure 6. It was determined that the algal standing crop and abrupt algal die-off, following treatment of the domestic and industrial wastes, will become the principal factor affecting the lake's dissolved oxygen. Figure 7 illustrates this relationship. Assuming an algal standing crop having a chlorophyll a concentration of 86 ug/l, observed for relatively clear waters in Onondaga Lake, a die-off could result in a DO concentration approaching 2.0 mg/l. Although phosphorus is likely to be the limiting nutrient, annual storm-related contributions of total phosphorus are negligible when compared to the projected dry-weather loads.

COST-EFFECTIVENESS ANALYSES

The development of relationships between CSO discharges and other wastewater sources relative to target water quality goals or standards is an important first step in assessing impacts and determining if CSO abatement is worth pursuing. Figure 8 presents an example of present and future dry-weather and the various wet-weather BOD loads in relation to a DO goal. If some degree of CSO abatement will result in a significant step toward a defined water quality goal, then the next task will be to perform a cost-effectiveness analysis.

A common economic approach to stormwater treatment has been to develop relationships between the impact of stormwater on receiving water quality and the cost of treatment for various storm conditions. This cost-benefit information has then been used by local governments and regulatory agencies to make reasonable economic decisions.

Figure 9 illustrates the interrelationships between the various phases of a CSO study and shows the development of a cost-effective treatment solution.

Figure 9A illustrates the realtionship between the pollutant loads discharged from a CSO system and various storm conditions described in terms of recurrence interval. Recurrence interval can be used in this manner only if other significant factors such as pollutant residual and ADW are held constant. Recurrence interval is an appropriate variable from the standpoint of the general availability of data and relationship to abatement costs.

Generally, several years of rainfall records are available whereby frequency of overflow can be determined on the basis of sewer system

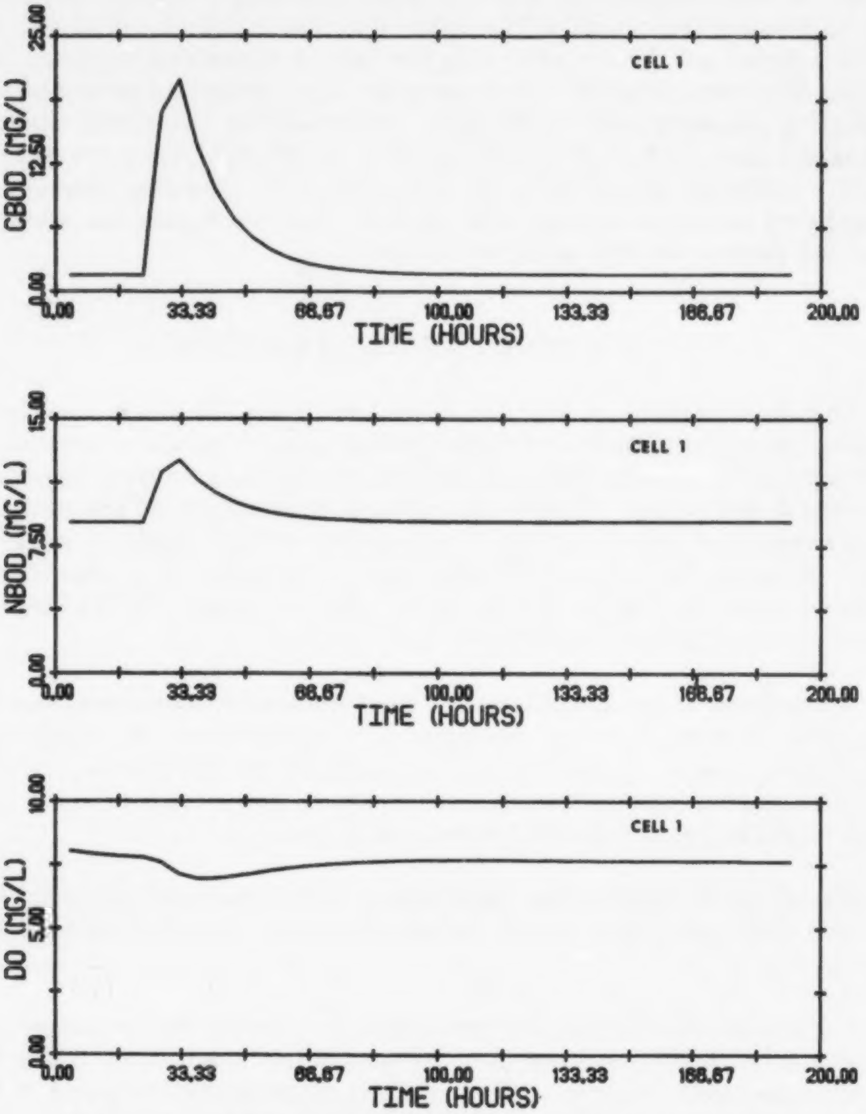


FIGURE 6. ONONDAGA LAKE PROJECTION FOR A ONCE IN TEN YEARS STORM: LAKE RESPONCE NEAR PRIMARY LOAD(18)

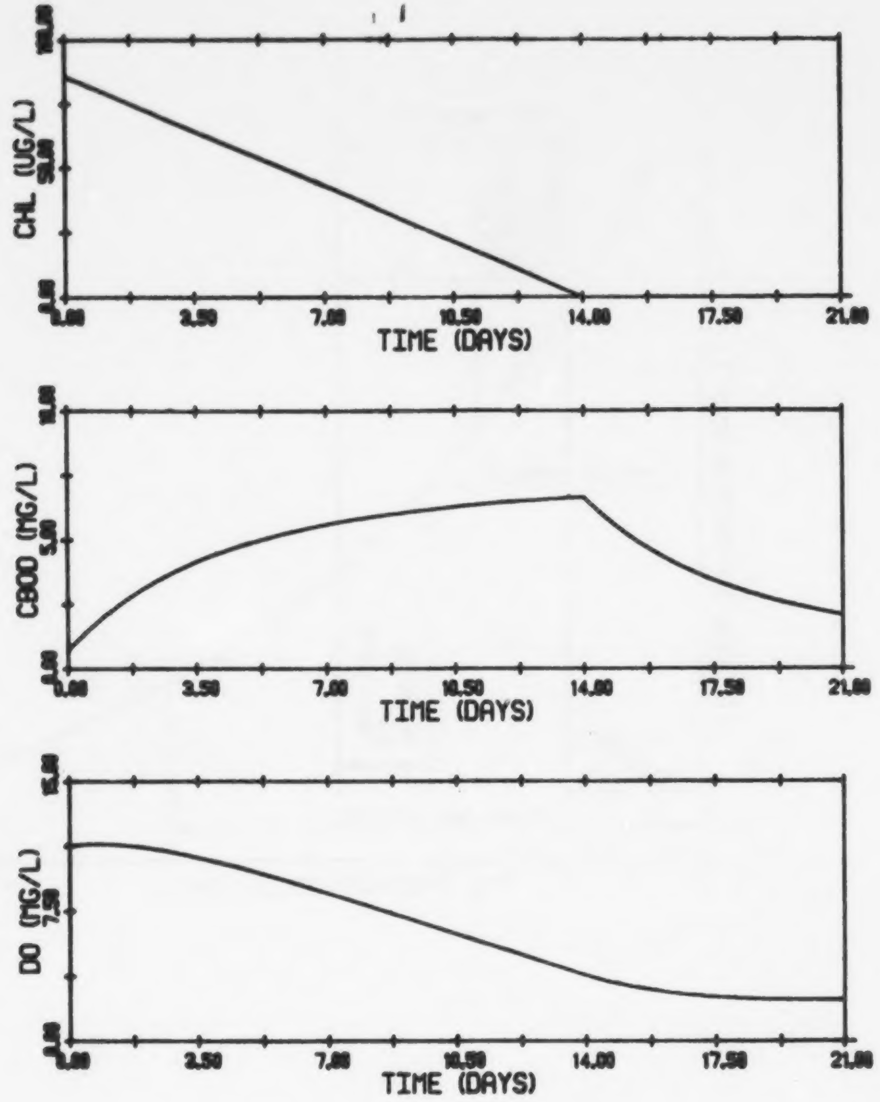


Figure 7. Model simulation of chlorophyll dieoff.(18)

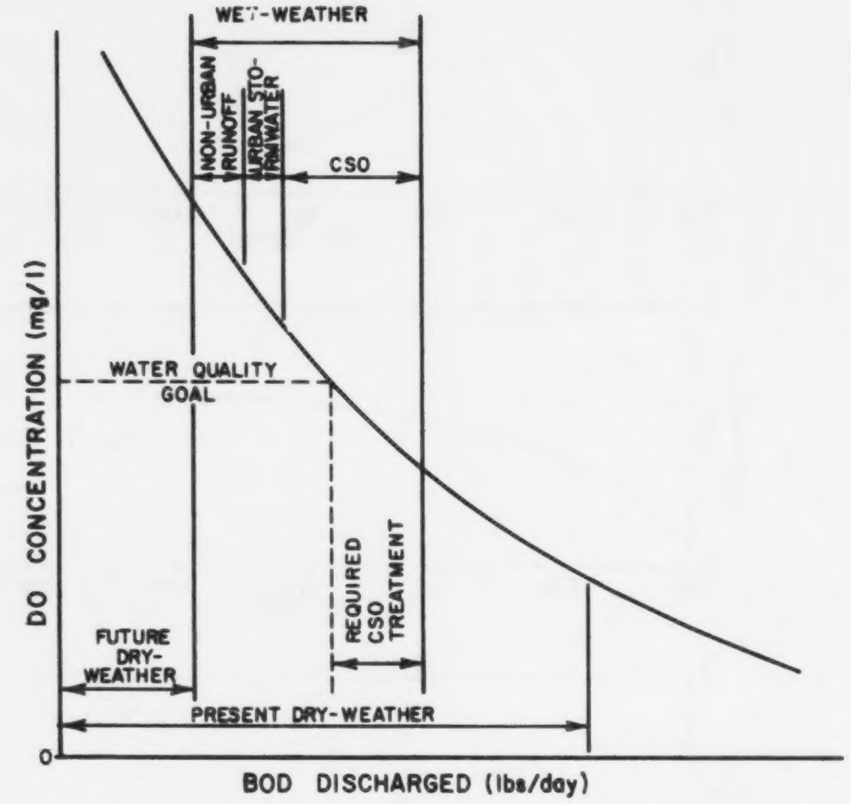
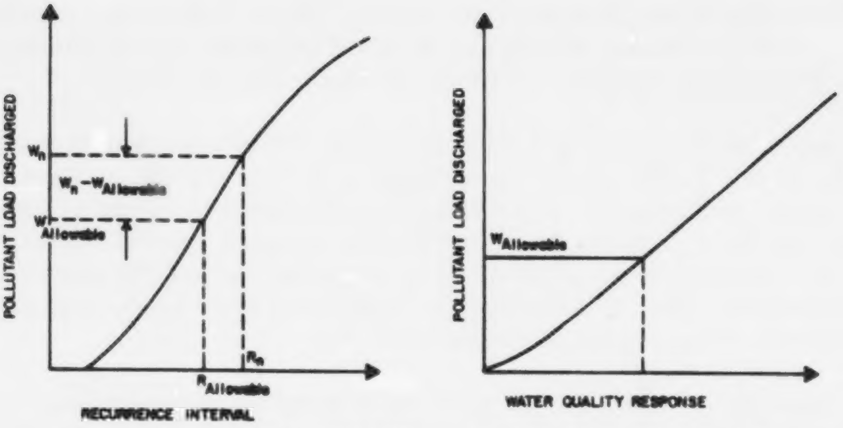
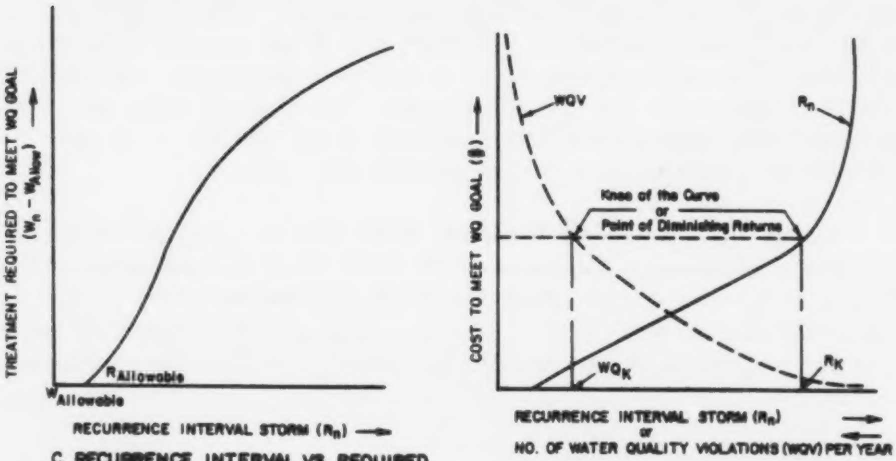


Figure 8. Determination of required CSO abatement relative to water quality goal.(4)



A. RAINFALL TYPE VS. POLLUTANT LOAD

B. POLLUTANT LOAD VS. WQ RESPONSE


TREATMENT REQUIRED TO MEET WQ GOAL ($W_n - W_{Allow}$) →
$R_{Allowable}$
$W_{Allowable}$
RECURRENCE INTERVAL STORM (R_n) →
COST TO MEET WQ GOAL ($) →
WQV
R_n
Knee of the Curve
or
Point of Diminishing Returns
WQ_K
R_K
RECURRENCE INTERVAL STORM (R_n) →
or
NO. OF WATER QUALITY VIOLATIONS (WQV) PER YEAR ←

C. RECURRENCE INTERVAL VS. REQUIRED TREATMENT

D. RECURRENCE INTERVAL VS. COST

Figure 9. Procedure for establishing the most cost-effective treatment level to meet a given water quality goal.(4)

capacity. Such information, in combination with even limited quality information, can be used to determine the pollutants discharged from a system over a period of years. This is accomplished either through a series of single-event calculations or continuous simulation.

The significant factors in developing costs of abatement strategies are flow rate and volume, which are derived from the characteristics of recurrence interval, namely, rainfall intensity and duration.

Figure 9B illustrates the relationship between pollutant load discharged from the sewer system and water quality response. Based on a selected water quality goal, the allowable pollutant load determined from Figure 9B can be related to an allowable recurrence interval from 9A. Although the relationships illustrated by Figures 9A and 9B can represent independent study efforts, they can be interfaced through the development of the common relationship to pollutant load.

Figure 9C illustrates the relationship between treatment required and recurrence interval. This relationship is based on a series of points selected from Figure 9B, and can be expressed in terms of the pollutant load in excess of the allowable (W_n-$W_{allowable}$). The treatment requirements set the basis upon which the cost of a treatment facilities can be determined. The various methods of source control and the various treatment options are reported elsewhere (9). The specific unit processes available and the associated unit costs for both construction and operation and maintenance are well documented (14, 15, 16).

Figure 9D illustrates the cost of treatment as it relates to both recurrence interval and the number of water quality violations. It should be noted that the costs represent the least expensive mix of control and treatment options in each case. These costs are developed on the basis of engineering judgment and relationships developed through previous studies (17, 18).

COST-EFFECTIVE SOLUTION

Treatment costs can be related directly to either recurrence interval or a measureable benefit, which in this case has been illustrated as the number of water quality violations per year. Since recurrence interval (R_n) serves as the basis for determining the frequency of overflow in the system, the number of water quality violations can be directly equated to recurrence interval. Another method to determine the number of water quality violations is through the use of continuous simulation, whereby the number of violations per year for a particular treatment scheme are calculated. Costs can then be plotted against the average annual benefit as shown in Figure 9D.

For the example selected, the knee of both curves shown occurs at the same cross point. The "knee" of the curves represents the point of diminishing returns. Below the knee, treatment may be as cost effective but will result in less removal and subsequently more frequent violations of the water quality goal. At all points beyond the knee, the additional benefits derived from incremental increases in expenditure decrease rapidly and are commonly referred to as being beyond the point of diminishing returns. While it may be desirable to meet water quality standards beyond the "knee", justification should be based on factors other than economics (e.g., legislative decisions or "best usage" of the receiving water).

A more comprehensive method of presenting abatement costs and associated benefits involves performing the above procedure for various water quality goals. Figure 10 shows a plot of costs versus water quality violations for various goals in order to illustrate the economic impact of more or less stringent goals. Such an approach can be used to determine the most cost-effective water quality goal or "general optimum solution" (GOS). The term GOS is used herein to denote a solution which encompasses the essential factors necessary to arrive at a reasonable solution. Figure 10 presents a summary of these factors from which the decision-makers can select a reasonable water quality objective.

The GOS can be determined by drawing a curve (A-B in Figure 10) through the knees of a series of cost curves similar to the one presented in Figure 9D. The general optimum solution to the cost-benefit analysis occurs at the knee of this curve. This point not only defines the water-quality standard and associated cost, but through the use of Figure 9D, indicates the probable frequency with which that standard will be contravened.

It should be noted that in some cases data will not yield a noticeable break or knee at which the GOS occurs. However, in any event, presentation of the data as shown in Figure 10 will be desirable since it provides the complete picture of the water-quality benefits which can be expected for any given expenditure. Even though a GOS exists, it may be desirable for a variety of reasons (e.g., limited funds or inability to live with the cost-effective water quality goal) to select another water quality goal. The cost data presented in Figure 10 enables the decision-makers to balance the water quality goals against the total cost of achieving them.

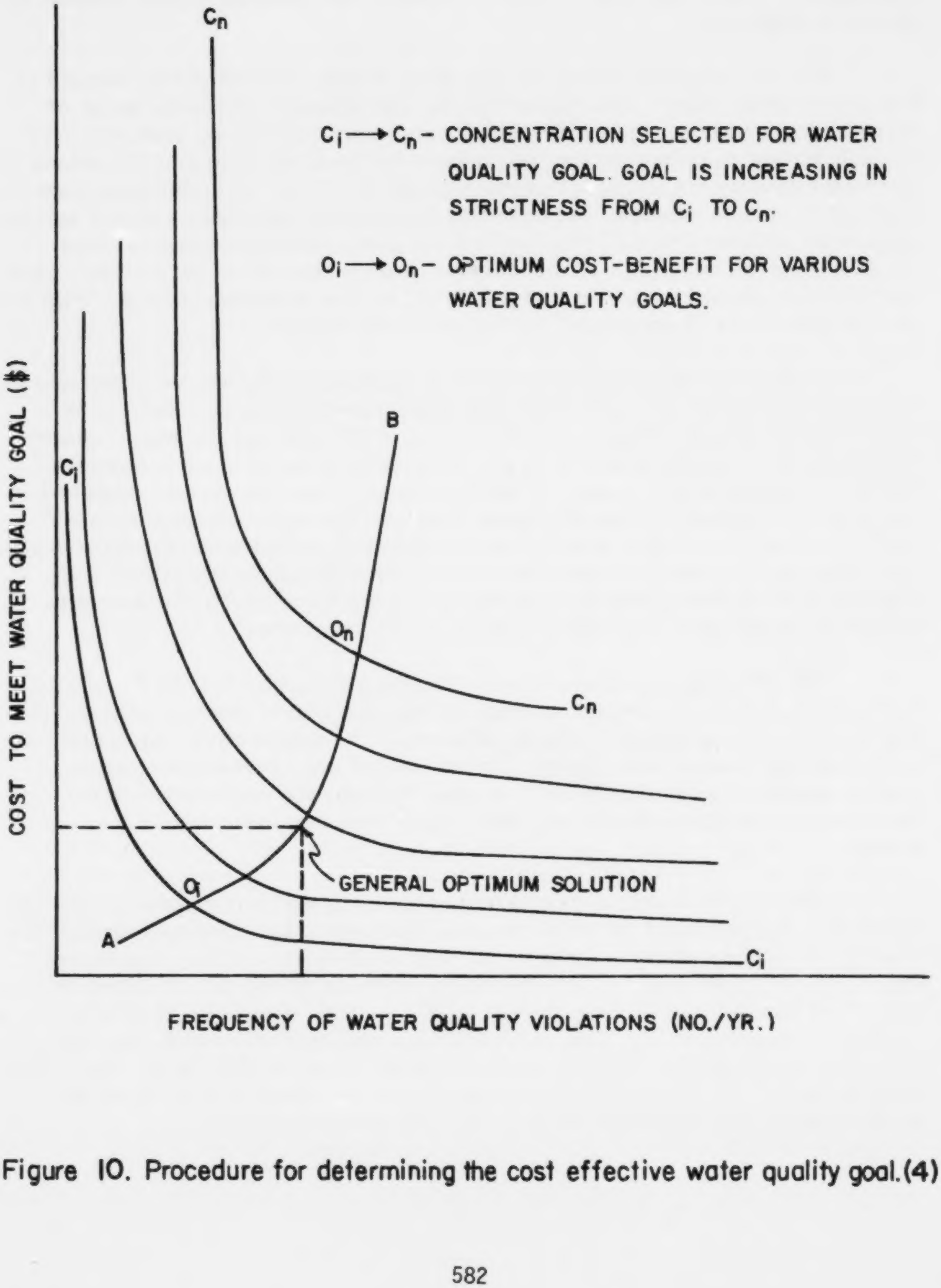


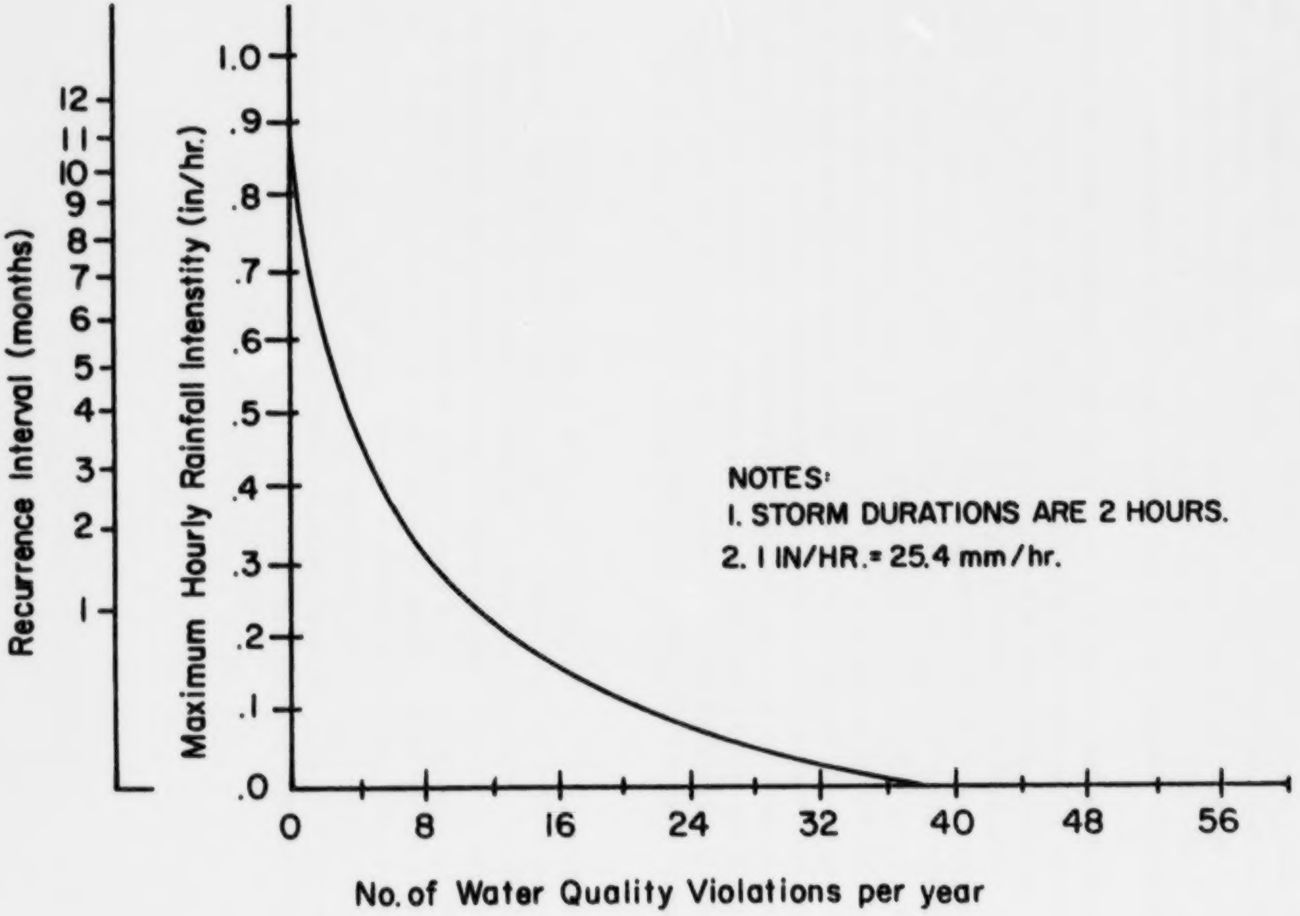
Figure 10. Procedure for determining the cost effective water quality goal.(4)

ONONDAGA COUNTY

Curves relating various recurrence-interval storms, CSO discharges and water quality were developed for the Onondaga Lake study. In that case, the analyses showed fecal coliform to be the only parameter to show significant water quality impact from the CSO discharges. Therefore, only examples of abatement solutions related to meeting the contact recreation standard of 200 cells/100 ml established by the New York State Department of Environmental Conservation will be discussed in this section.

Extensive statistical analyses of rainfall patterns were performed on the historical records for Syracuse, New York, and the results compared to the findings of both the CSO and the water quality programs. This procedure revealed that there are 170 rainfall events per year; based on peak intensity, 65 of these produce overflows. On the average, the combined sewer system is capable of accepting the runoff from a storm with peak intensity of less than 0.05 inches/hour without causing any overflow. Of these 65 overflow-producing events, only 38 result in discharges of sufficient magnitude to cause a violation of the fecal coliform standard in the contact recreation zone. Figure 11 presents the number of violations that occur annually as a result of treating storms of various intensities or recurrence intervals for a fixed two-hour duration. Figure 12 presents a similar plot in terms of days of violation. Figure 13 shows the present worth costs (1983) for constructing abatement facilities to treat these storms.

Inspection of the curve presented in Figure 13 shows the "knee" of the curve occurring in the vicinity of $12 million; even with that investment, 13 violations would occur per year. Of the 13 annual violations, over ten will occur during the recreation season from June through September as illustrated by Figure 14. This case is one where the knee-of-the-curve solution is unacceptable. Since ten violations would essentially prohibit use of the lake for contact recreation, more stringent levels of treatment will ultimately be needed. The selected goal, for which treatment will eventually be provided, is to allow one violation of the contact recreation standard (200 cells/100 ml) per year.



NOTES:
1. STORM DURATIONS ARE 2 HOURS.
2. 1 IN/HR. = 25.4 mm/hr.

Figure 11. Number of water quality violations occurring annually in Onondaga Lake as a result of various intensity storms (5)

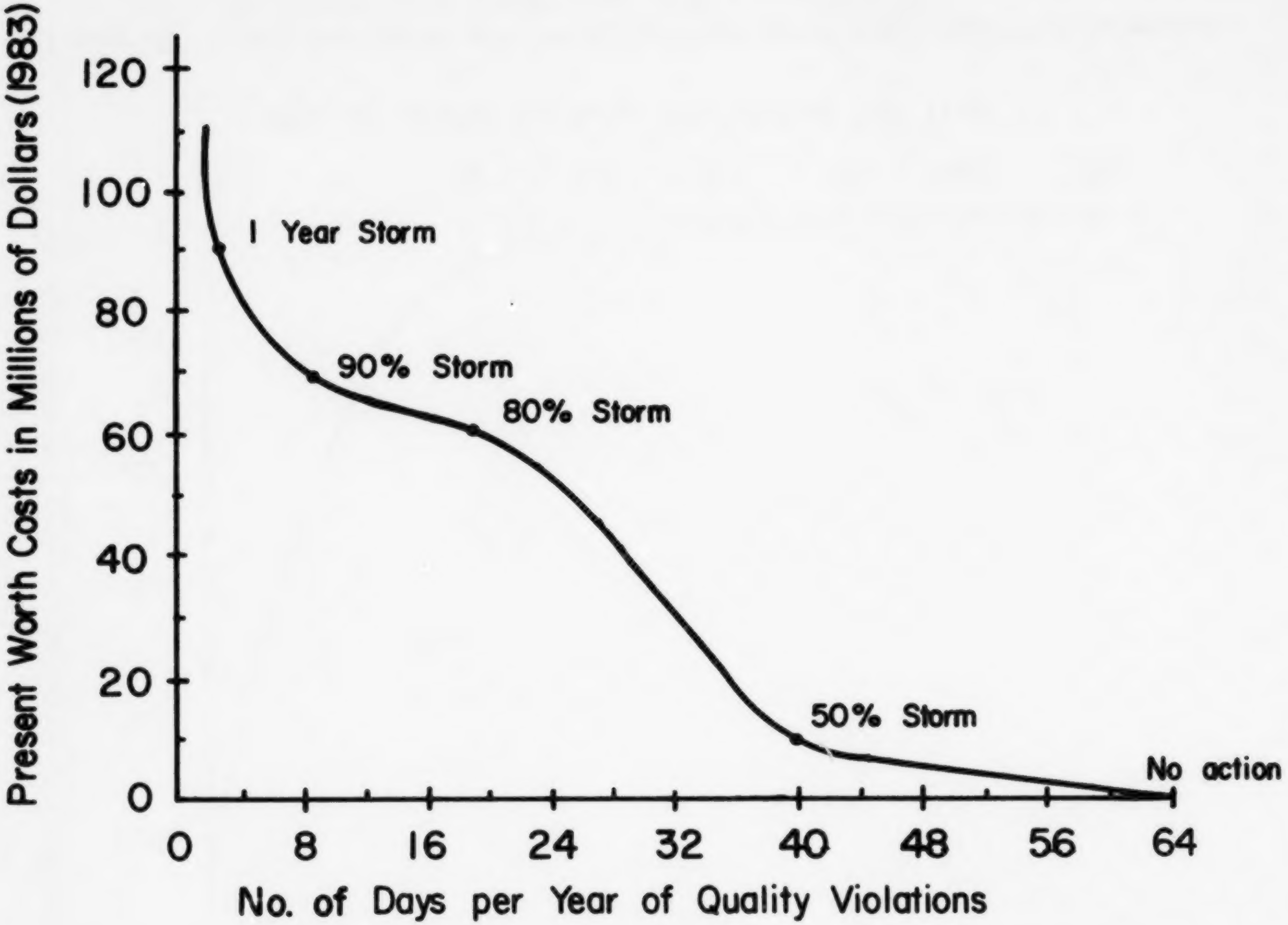


Figure 12. Costs associated with treatment of combined sewer overflows to reduce the frequency of water quality violations. Onondaga County(5)

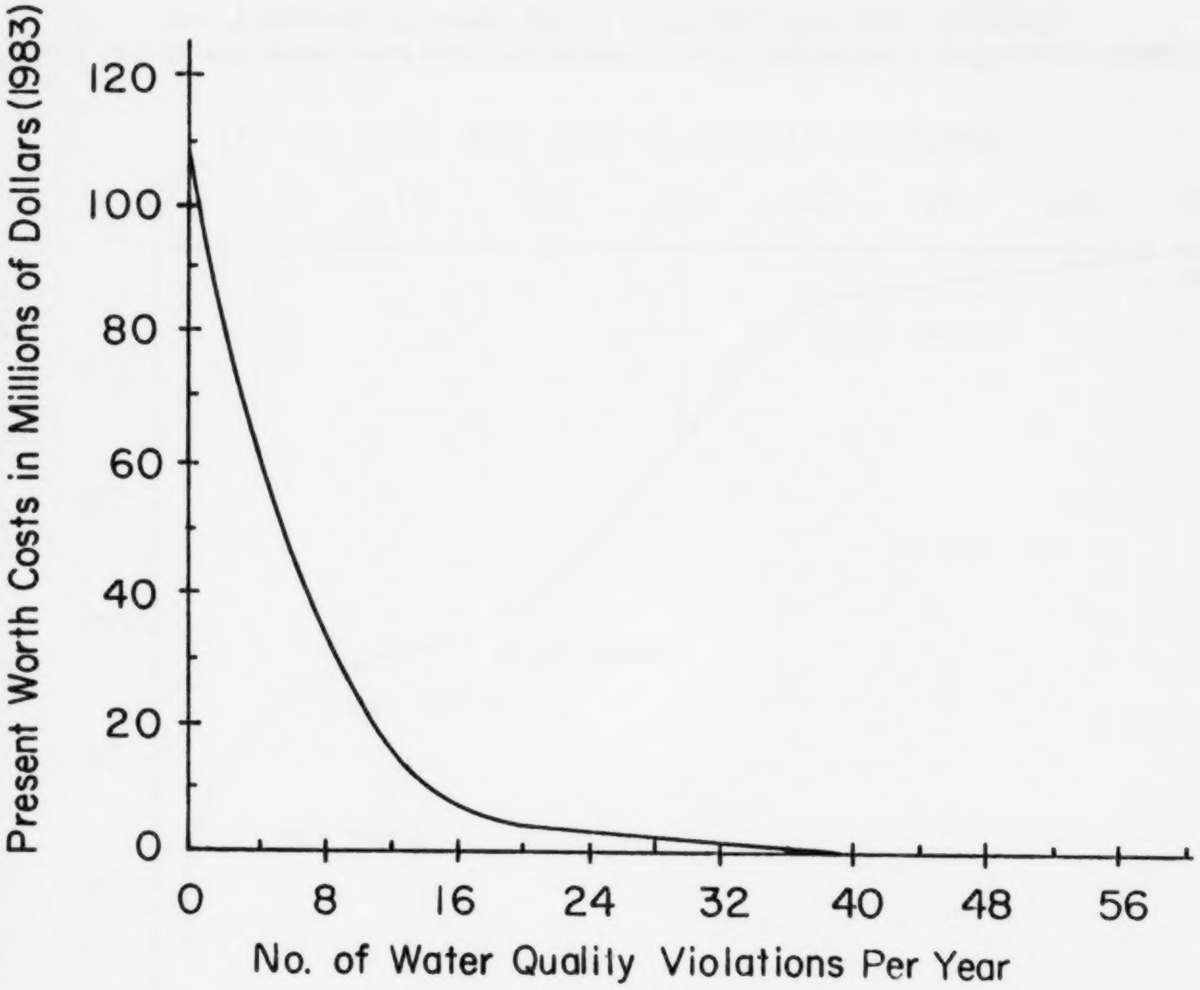
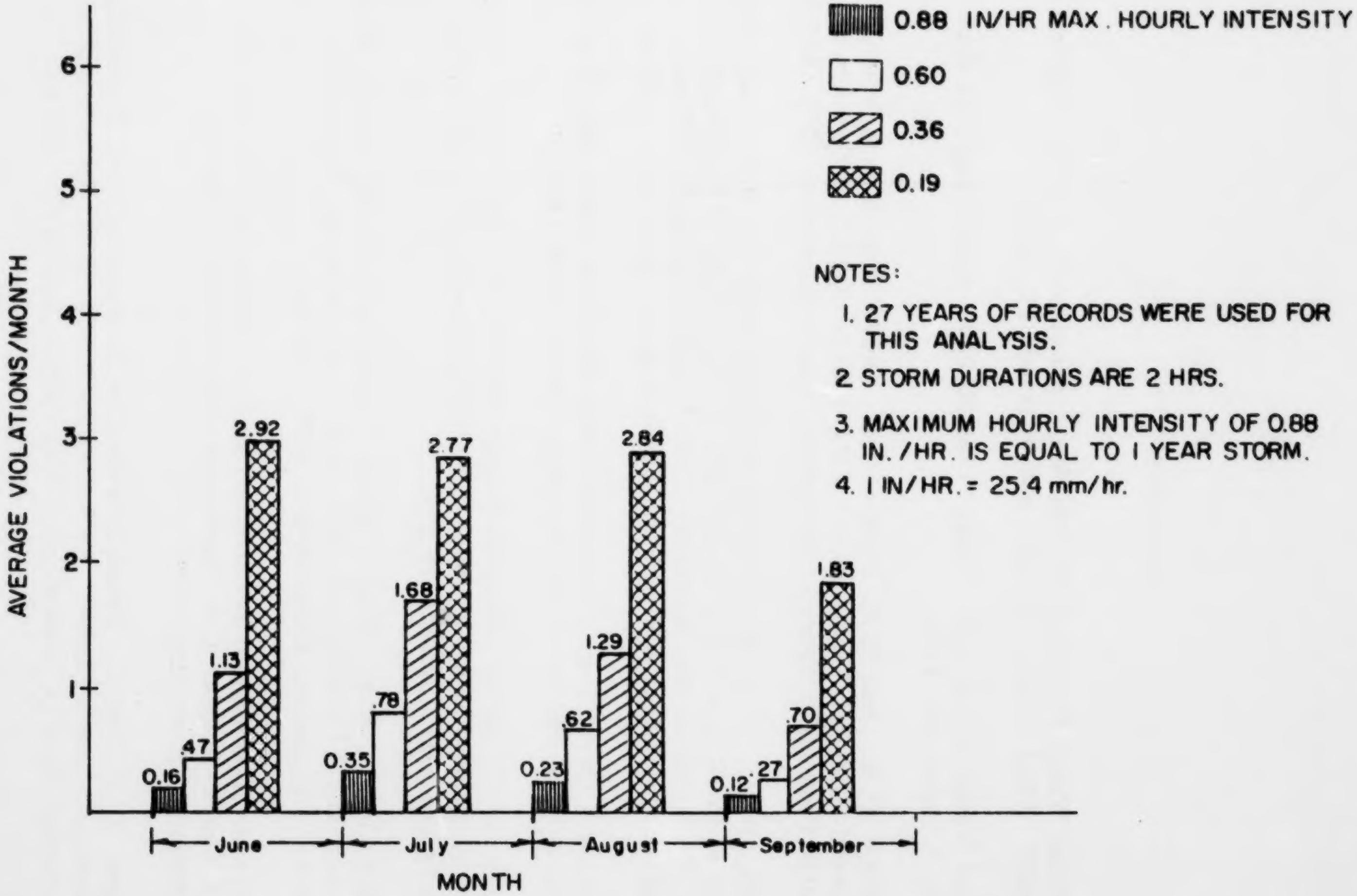


Figure 13. Costs associated with treatment of combined sewer overflows to reduce the frequency of annual water quality violations. (5)



NOTES:

1. 27 YEARS OF RECORDS WERE USED FOR THIS ANALYSIS.
2. STORM DURATIONS ARE 2 HRS.
3. MAXIMUM HOURLY INTENSITY OF 0.88 IN./HR. IS EQUAL TO 1 YEAR STORM.
4. 1 IN/HR. = 25.4 mm/hr.

Figure 14. Average number of violations of fecal coliform standards during the recreation season resulting from various intensity storms. Onondaga Lake, New York (5).

REFERENCES

1. Federal Water Pollution Control Act Amendments of 1972, Public Law 92-500. 92nd Cong., 86 Stat. 816, October 18, 1978.

2. Clean Water Act of 1977, Public Law 95-217. 95th Cong., 91 Stat. 1566, October 27, 1977.

3. Shelley, P.E., and G.A. Kirkpatrick. Sewer Flow Measurement, A State-Of-The-Art Assessment. EPA-600/2-75-027, U.S. Environmental Protection Agency, Cincinnati, Ohio, November 1975. 436 pp.

4. Moffa, P.E., et al. Methodology for Evaluating The Impact and Abatement of Combined Sewer Overflows, A Case Study of Onondaga Lake, New York. Grant No. R805096, U.S. Environmental Protection Agency, Cincinnati, Ohio, Final Draft Report, June 1979. 120 pp.

5. O'Brien & Gere, Inc. Combined Sewer Overflow Abatement Program. Report Being Prepared for Onondaga County, New York.

6. Huber, W.C., et al. Interim Documentation, November 1977 Release of EPA SWMM, Draft Report. U.S. Environmental Protection Agency, Cincinnati, Ohio, November 1977.

7. Huber, W.C., et al. Storm Water Management Model User's Manual Version II. EPA-670/2-75-017, U.S. Environmental Protection Agency, Cincinnati, Ohio, March 1975. 367 pp.

8. Areawide Assessment Procedures Manual, Volumes I, II, and III. EPA-600/9-76-014, U.S. Environmental Protection Agency, Cincinnati, Ohio, July 1976.

9. Lager, J. A., et al. Urban Stormwater Management and Technology: Update and Users' Guide. EPA-600/8-77-014, U.S. Environmental Protection Agency, Cincinnati, Ohio, September 1977. 331 pp.

10. Amy, G., et al. Water Quality Management Planning for Urban Runoff. EPA-440/9-75-004, U.S. Environmental Protection Agency, Washington, D.C., December 1974.

11. U.S. Army Corps of Engineers. Storage-Treatment, Overflow, Runoff Model "STORM" User's Manual. Hydrologic Engineering Center, Davis, California, July 1976.

12. Lager, J.A., et al. Development and Application of a Simplified Stormwater Management Model. EPA-600/2-76-218, for U.S. Environmental Protection Agency, Cincinnati, Ohio, August 1976. 153 pp.

13. Limno-Tech, Inc. Mathematical Modeling of the Impact of Storms on Water Quality in Onondaga Lake. Prepared for Stearns & Wheler, Civil and Sanitary Engineers, and Onondaga County, New York. Ann Arbor, Michigan, 1978. 133 pp.

14. Heaney, J.P., and S. J. Nix. Storm Water Management Model: Level I - Comparative Evaluation of Storage-Treatment and Other Management Practices. EPA-600/2-77-083, U.S. Environmental Protection Agency, Cincinnati, Ohio, April 1977. 105 pp.

15. Reid, G.K. Ecology of Inland Waters and Estuaries. Reinhold Publishing Corporation, New York, New York, 1961. 375 pp.

16. Turner, B.G., et al. 1976 Needs Survey. EPA-430/9-76-012, U.S. Environmental Protection Agency, Washington, D.C., February 1977. 400 pp.

17. Sonnen, Michael B. Abatement of Deposition and Scour in Sewers. EPA-600/2-77-212, U.S Environmental Protection Agency, Washington, D.C., November 1977. pp. 115.

18. Stearns & Wheler, Civil and Sanitary Engineers. Onondaga Lake Storms Impact Study. Report Prepared for Onondaga County. Cazenovia, New York, August 1979.

A WATER QUALITY PLANNING METHODOLOGY FOR URBAN AREAS

Franklin W. (Skip) Ellis
CH2M HILL, Reston, Virginia
Ronald L. Wycoff
CH2M HILL, Gainesville, Florida

Abstract

The Environmental Protection Agency's (EPA) 1978 Needs Survey reported that an estimated $36 billion is required to construct necessary municipal wastewater treatment facilities. Additionally, $26 billion is required to control pollution from combined sewer overflow, and $46 billion is required to control urban stormwater runoff. In light of the current trent toward reduced taxes, local funds for water quality projects will become more limited. Moreover, the public should demand that such projects demonstrate perceptible improvements in water quality and are required for the protection of the intended beneficial use of the receiving water. Further, the most cost-effective pollution control alternatives should be used. Without this assurance required pollution abatement projects may well go unfunded.

Based upon information gained in the performance of the EPA's 1978 Needs Survey, a two-phase approach to water quality planning is presented that determines the most cost-effective mix of control alternatives and their impacts on the receiving water. This approach is general in nature but is oriented toward combined sewer overflow (CSO) areas and the requirements of EPA's Program Requirements Memorandum, PRM No. 75-34.

Phase I is an initial assessment designed to answer the following questions:

1. What is the intended beneficial use of the receiving water?
2. What water quality goals or criteria are required to ensure this use?
3. Based on the goals, is there a water quality problem?
4. Can CSO, urban runoff, or domestic wastewater abatement techniques solve the problem, and what degree of controls is required?
5. If the problem can be solved, what controls, in general, are the most cost effective, and what is the nature of the tradeoffs between cost, degree of control, and receiving water quality?
6. Considereing these tradeoffs, are changes in the water quality goals, or in the desired beneficial use for the receiving water indicated?

This two-phase approach is designed to achieve water quality goals established to protect a given beneficial use of the receiving water. Rather than a fixed limit, such as a minumum dissolved oxygen standard, statistical water quality responses are used as the goals. Examples from the EPA's 1978 Needs Survey illustrate this point.

Phase I involves the use of a simple, continuous receiving water quality model and the Heaney-Nix economic optimization procedure. The model, the Continuous Stormwater Pollution Simulation System (CSPSS), incorporates the pertinent features of an urban area and its receiving water. A very brief description of CSPSS is given. The economic optimization is performed using estimated areawide production functions for a first-cut analysis of the costs required for various levels of pollution abatement. The water quality model uses these relationships to produce cost-water quality relationships that can be used to guide planners as to the desirability of various pollution abatement projects. Studies from areas with CSO's are presented.

If the results of the Phase I analysis indicate that there are water quality problems that can be corrected with affordable solutions, the detailed analysis of Phase II is required. This phase considers, in much greater detail, the area's hydrology, combined sewer system hydraulics, nonpoint and point source pollutants, and the receiving water quality response, thus requiring more sophisticated data and models.

The product of the Phase II analysis is a description of the optimal mix of control alternatives, the total plan costs, and the receiving water quality response due to the plan. The description of the alternatives includes the level of effort required, the area to which the alternative applies, the expected pollutant reduction due to the alternative, and the cost of the alternative.

In summary, this two-phase approach can result in substantial monetary savings by obtaining economically optimal solutions. Use of this methodology also explicitly evaluates the benefits or improvements in water quality as a result of the project. These features satisfy EPA planning requirements and should assure the public that their tax dollars are wisely spent.

INTRODUCTION

The Environmental Protection Agency's (EPA) 1978 Needs Survey[1] estimates that $36 billion will be required to construct necessary municipal wastewater treatment facilities, including secondary treatment and AWT plants. In addition, $26 billion will be required to control pollution from combined sewer overflow (CSO) and $62 billion will be required to control stormwater runoff (SWR) pollution from existing urbanized areas. Considering the current nationwide trend toward reduced taxes, local funds for water quality projects will become more limited. The public will demand that pollution control projects competing for dwindling tax dollars demonstrate perceptible improvements in water quality and use the most cost-effective pollution control alternatives. Without this assurance, beneficial projects may go unfunded.

Program Requirements Memorandum PRM 75-34 (Program Guidance Memorandum PG-61), issued by the EPA, requires that projects funded under the construction grants program for control of pollution from combined sewer overflow be both cost effective and necessary for the protection of the beneficial use of the receiving water. PRM 75-34 also provides several other criteria which should be met before potential CSO control projects are funded. These are:

1. Provision has already been made for funding secondary treatment of dry-weather flows.

2. The pollution control technique proposed for CSO is a more cost-effective method of protecting the beneficial use of the receiving water than other CSO control techniques or the addition of wastewater treatment higher than secondary for dry-weather municipal flows.

3. The marginal costs are not substantial compared to the marginal benefits.

In addition to the above requirements for CSO control, a more recent (May 1979) Program Requirements Memorandum (PRM 79-7) has established a case-by-case EPA-headquarters-level review procedure for all significant projects providing municipal wastewater treatment more stringent than secondary. In order to be approved for funding, these projects must demonstrate significant water quality and public health improvements. Requirements such as these have been developed to ensure cost-effective and water-quality-effective allocation of limited construction grants funds.

A two-phase approach to urban water pollution control planning which achieves the goals of PRM 75-34 and PRM 79-7 is presented. The approach determines the degree of control required, the most cost-effective mix of water quality management alternatives, and the impacts of these alternatives on the receiving water. The methodology can accommodate staged implementation, allowing less stringent water quality goals to be achieved first while the option of achieving more stringent goals in the future is retained. The methodology will also provide a clear understanding of the actual costs incurred in meeting water quality goals and/or standards, "all of the time" versus the cost incurred when an allowable frequency of exceedance of the standard is accepted. This provides a feedback system for evaluating the economic practicality of meeting certain water quality goals.

PHASE I - INITIAL ASSESSMENT

The first phase of the cost-effective water quality planning approach is designed as an initial assessment of water quality problems and potential solutions. The methodology, based in part upon information and experience gained in the performance of the Categories V and VI portion of the 1978 Needs Survey,[2] is intended to answer the following questions:

1. What is the intended beneficial use of the receiving water?
2. What water quality goals or criteria are required to ensure this use?
3. Based on these goals, is there a water quality problem?

If the answer to question 3 is no, then the planning process is complete and additional water quality control facilities are not required. If the answer to question 3 is yes, then the following questions must be addressed:

4. Can combined sewer overflow, urban runoff, or additional point source pollution abatement techniques solve the problem; and what degree of control is required?
5. If the problem can be solved, which controls, in general, are the most cost effective?

Responding to questions 4 and 5 will result in the formulation of a pollution control project as well as a Preliminary Cost Analysis. If the project is readily affordable and the water quality goals are fully met, then the analyses can proceed to a more detailed study, i.e., Phase II. However, if the costs are excessive or the water quality goals cannot be met, the following additional questions should be addressed:

6. What is the nature of the tradeoffs between cost, degree of control, and receiving water quality?
7. Considering these tradeoffs, are changes in the water quality goals or in the desired beneficial use of the receiving water indicated?

If the answer to question 7 is yes, then new water quality goals and/or beneficial uses must be established and the results of the Phase I planning process reevaluated until both acceptable goals and affordable solutions are obtained.

The initial Phase I assessment involves the determination of the beneficial uses of the receiving water and the establishment of appropriate water quality goals or criteria in order to answer questions 1 and 2. Continuous water quality data and/or a continuous water quality computer simulation model are used to determine the nature, sources, and extent of water quality problems, a procedure which will answer question 3. The model is also used to obtain a preliminary estimate of the degree of pollution abatement that would be required to achieve the water quality goals, thus answering question 4. An economic optimization procedure is used to determine the most cost-effective mix of control alternatives for various levels of pollution abatement, which answers question 5 and 6. Finally, a feedback process allows decisionmakers a chance to evaluate the proposed project in light of costs and receiving water response, which provides the necessary information to answer Question 7.

RECEIVING WATER QUALITY GOALS

Water quality goals or criteria generally vary according to the intended beneficial use of the receiving water and act as an indicator for evaluating whether water quality problems exist. Water quality criteria are also used in the determination of the level of required pollution abatement if a water quality problem is identified.

Water quality goals are often defined for a number of pollutants or water quality indicators. However, for the purpose of this paper, only dissolved oxygen (DO) will be considered. The methodology outlined here is general and may be applied to other parameters.

The intent of the Clean Water Act is to achieve fishable and swimmable waters. The concentration of dissolved oxygen in a receiving water is one of the most important indicators of the ability of the water to support a viable fishery. Urban areas can supress the DO concentration by the introduction of oxygen-demanding substances from industrial waste discharges, municipal wastewater treatment plant discharges, combined sewer overflow, and stormwater runoff. The combined effects of these sources may be evaluated by considering the DO budget of the receiving water.

The dissolved oxygen criteria proposed by EPA[3] for freshwater fish is:

> Freshwater aquatic life: A minimum concentration of dissolved oxygen to maintain good fish populations is 5.0 mg/l. The criterion for salmonid spawning beds is a minimum of 5.0 mg/l in the interstitial water of the gravel.

Most states have established fishery standards that basically agree with these criteria. Many states differentiate between cold-water fish (salmonids) and warm-water fish by setting daily average cold-water fish DO limits of 6.0 mg/l and warm water limits at 5.0 mg/l. These criteria are the lower limits of DO concentrations that fish can tolerate for extended periods of time. However, DO levels as low as 2.0 to 3.0 mg/l can be occasionally tolerated for short periods without causing fish kills. Based on this fact, a minimum DO level and its associated allowable frequency of exceedance were developed as a basis for estimating facilities needs required for control of combined sewer overflow and urban stormwater runoff in the 1978 Needs Survey.[2]

From available data on juvenile Brook Trout[4,5] a relationship between lethal and safe DO concentrations and exposure times was developed and is shown on Figure 1. Since juvenile Brook Trout are an extremely sensitive species, Figure 1 should be conservative for most receiving waters.

The selected DO criteria used to evaluate combined sewer overflow (Category V) and urban stormwater runoff (Category VI) facilities required to provide a viable fishery are given as follows:

1. The minimum receiving water dissolved oxygen concentration shall not average less than 2.0 mg/l for more than 4 consecutive hours.
2. The minimum receiving water dissolved oxygen concentration shall not average less than 3.0 mg/l for more than 72 consecutive hours (3 days).

3. The annual average receiving water dissolved oxygen concentrations shall be greater than 5.0 mg/l for all waters which support warm-water species and shall be greater than 6.0 mg/l for all waters which support cold-water (Salmonid) species.

Results of the simulation studies developed in the 1978 Needs Survey indicate that if the 2.0 mg/l for 4 consecutive hours criterion is met, then in general all other criteria are met. That is, this criterion in most cases controls the level of BOD removal required. The above criteria are used as the receiving water quality goals in the case studies portion of this paper.

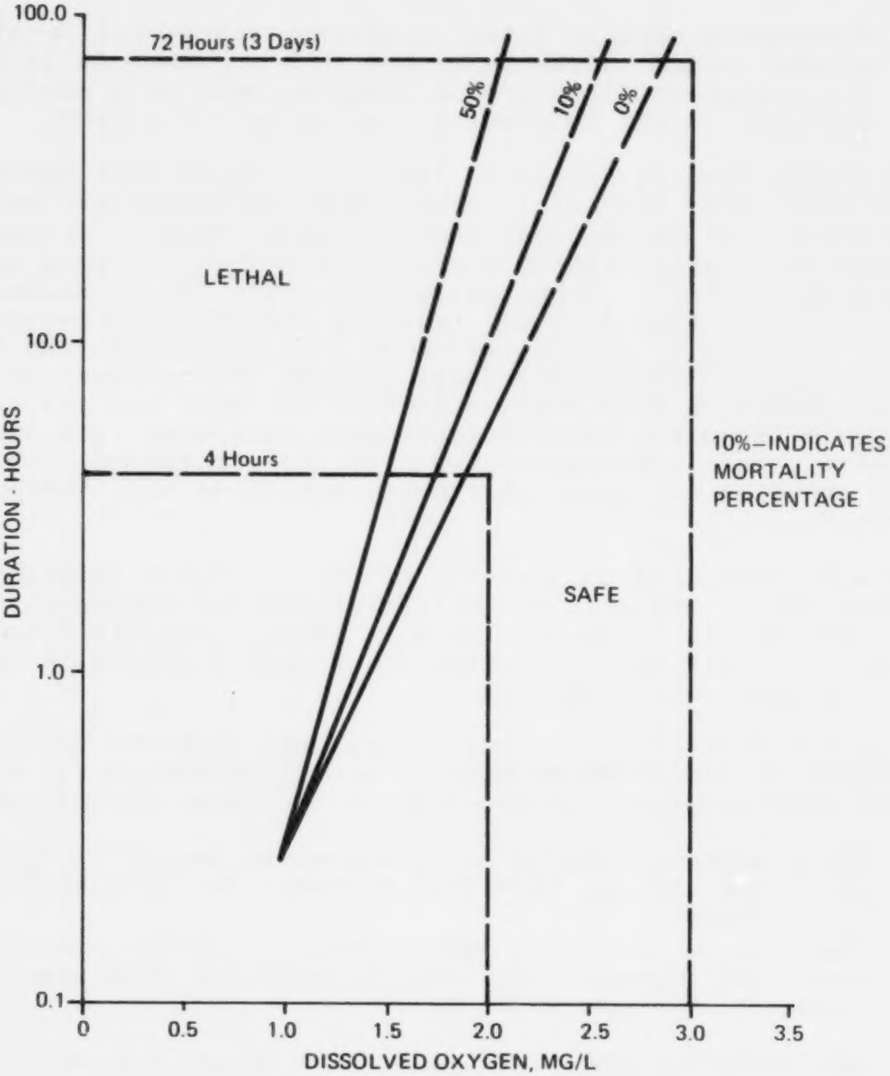


FIGURE 1. Mortality of Juvenile Brook Trout due to low DO levels.

These criteria are an example of identification of the intended beneficial use of the receiving water (i.e., establishment of a viable fishery) and the development of a quantitative statement of the water quality goals required to ensure this use. Necessarily, these are the first two steps of the water quality planning process.

CONTINUOUS WATER QUALITY SIMULATION

Evaluation of the compliance or noncompliance of receiving water quality with a desired water quality goal requires a long-term record (several years) of water quality conditions at one or more points on the receiving stream. Such records are generally unavailable; however, properly calibrated water quality computer simulation models can be used for the problem analysis. The time variant nature of receiving water quality requires the use of a continuous model. In addition, the model needs to consider point and nonpoint sources, upstream flow, upstream water quality, and the dynamics of the dissolved oxygen consumption and reaeration process. For the purposes of a preliminary analysis, a relatively simple model that is inexpensive to operate is desired.

Model Description

A model fulfilling these requirements was developed for the 1978 Needs Survey. The Continuous Stormwater Pollution Simulation System (CSPSS) is described in detail by Wycoff and Mara;[6] only a brief summary will be given here. The purpose of CSPSS is to simulate all major urban pollution sources and the receiving water response in a simple yet rational manner. Basic functions which may be simulated on a continuous basis are (1) local rainfall, (2) local runoff, (3) pollutant washoff, (4) sewer system infiltration, (5) storage/ treatment systems, (6) dry-weather wastewater flow, (7) receiving water streamflow, and (8) receiving water quality response. These modules may be executed in logical sequential order (shown on Figure 2) to produce the desired simulation.

The model determines the water quality response of the receiving stream immediately downstream from an urban area. All waste sources, including urban stormwater runoff, combined sewer overflow, wastewater treatment plant effluent, and upstream flow, are considered. Oxygen demands included are ultimate carbonaceous BOD, nitrogenous BOD, sediment demand, and background dissolved oxygen deficit. The only oxygen source considered is atmospheric reaeration.

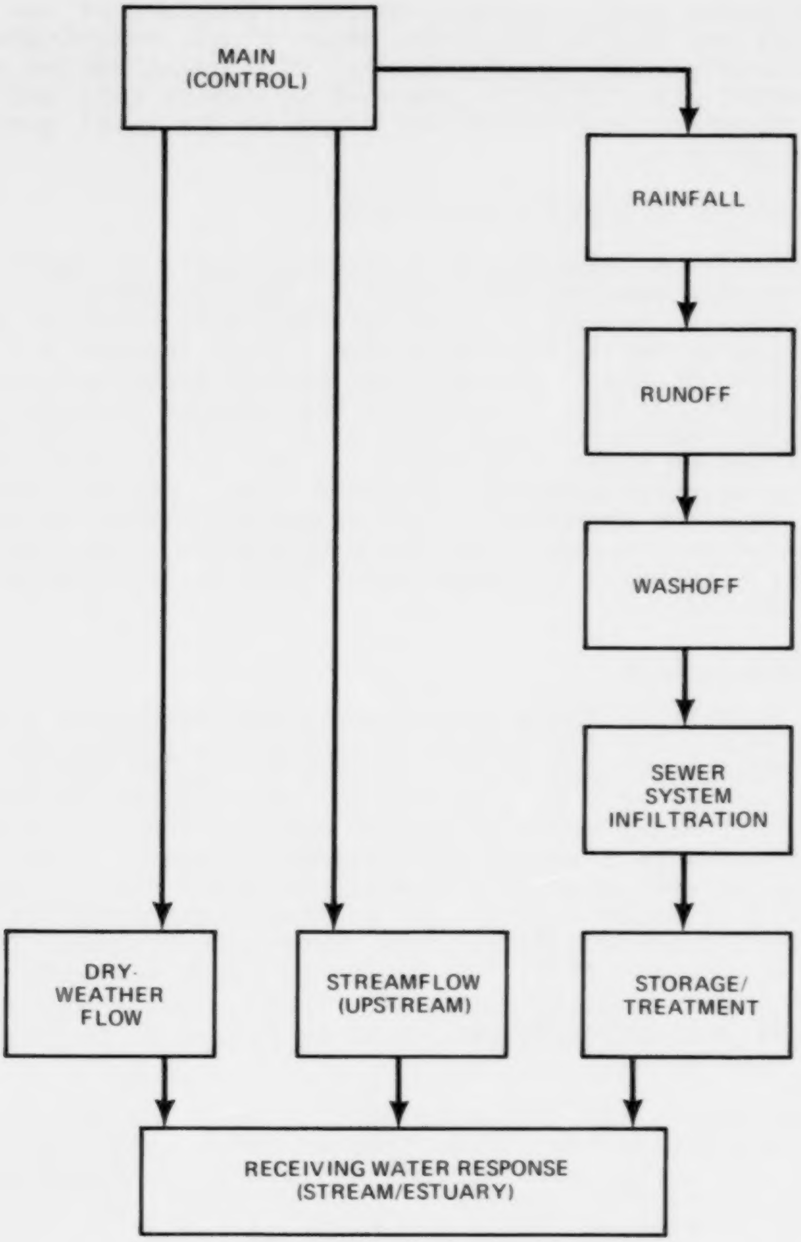


FIGURE 2. General flow chart for CSPSS.

The model generates the data required to construct a cumulative frequency distribution curve for each water quality parameter considered, including minimum receiving water DO concentrations for both existing and proposed conditions.

Water Quality and Alternative Assessment

The results of the water quality model simulation or observed data can be used to determine whether the water quality goals are met. If the goals are not met, then the severity of the water quality problem can be ascertained. Figure 3 shows a hypothetical example of a cumulative frequency curve for existing conditions and for a proposed improvement to current pollution abatement controls. For this case, simulation of the existing conditions shows that the 5.0-mg/l DO level is exceeded 30% of the time and 2.0-mg/l DO level is exceeded 2% of the time. If the DO criteria previously presented are applied in this case, then a water quality problem is identified.

The effects of improved point and nonpoint source controls can be evaluated by CSPSS to determine whether the water quality goals can be achieved and what degree of control is required. Simulation of a proposed pollution abatement control shows (Figure 3) that the 5.0-mg/l limit is exceeded only 2% of the time and the 2.0-mg/l limit is no longer violated, indicating substantial improvement in water quality. CSPSS can be used to determine the minimum degree of pollution abatement required to meet the water quality goals, hence avoiding costly overdesign of control alternatives.

If a model such as CSPSS has been calibrated to an urban area receiving water system, then the effect of reducing each controllable pollution source on receiving water quality can be investigated. This is accomplished with multiple simulations, each representing a different pollutant loading condition. The suggested procedure is to run simulations with all major pollutant sources held constant except one, which is varied. Once a series of runs are obtained and the results are analyzed, the effect on receiving water quality of reductions in pollutants from that source is known. This process is repeated until pollutant reduction/water quality response relationships are established for each controllable pollution source. These relationships, plus pollution control alternative cost data, become input to the economic optimization portion of the water quality planning process.

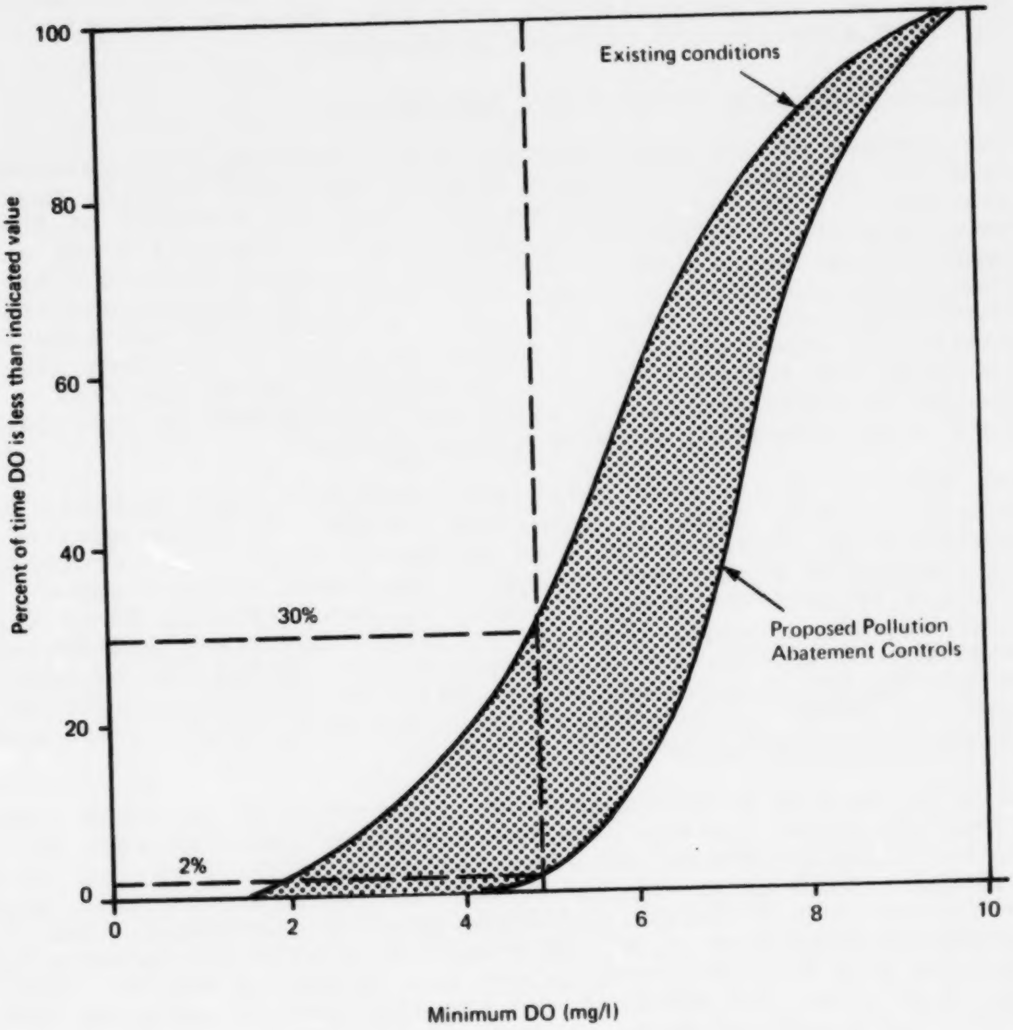


FIGURE 3. Example of cumulative frequency curves for minimum dissolved oxygen.

ECONOMIC OPTIMIZATION

Optimization is the process of making a system as perfect, effective, or functional as possible. Economic optimization is therefore the process of maximizing a system's output or production while minimizing its cost. In pollution control, economic optimization is the process of minimizing the costs required to achieve a certain water quality objective. Thus, the purpose of the economic optimization portion of the water quality planning process is to identify the technology or mix of technologies which will achieve the water quality objectives at the least possible total cost.

Production theory and marginal cost analysis as described by Heaney and Nix[7] can be used to identify the least costly combination of point and nonpoint source pollution control alternatives. In pollution control, a production function is the mathematical relationship between the amount of pollutant removed or the improvement in receiving water quality and the level of effort applied. The definition of level of effort depends on the particular control measure. For example, the level of effort for streetsweeping has been defined as the fraction of days that streets are swept, while for sewer flushing it has been defined as the fraction of sewers flushed daily. The shape of the production function is governed by the "law" of diminishing returns, which states that as an input to a production process is increased with all other inputs held constant, a point will be reached beyond which any additional input will yield diminishing marginal output.

In economic decisionmaking, marginal analysis determines whether an action results in a sufficient additional benefit to justify the additional cost. It indicates that more intensive use should be made of control alternatives with lower marginal costs. As these activities are expanded, marginal costs increase to the point where other options may become competitive. If this occurs, then the optimal (least-cost) solution to the pollution control problem will include a mix of controls, each utilized in an economically optimum manner.

Figure 4 summarizes the economic optimization procedure for a watershed served by a combined sewer system. Two nonstructural (or low structural) control technologies, streetsweeping (SW) and sewer flushing (SF), are considered along with a previously optimized storage/treatment system. The nonstructural controls,

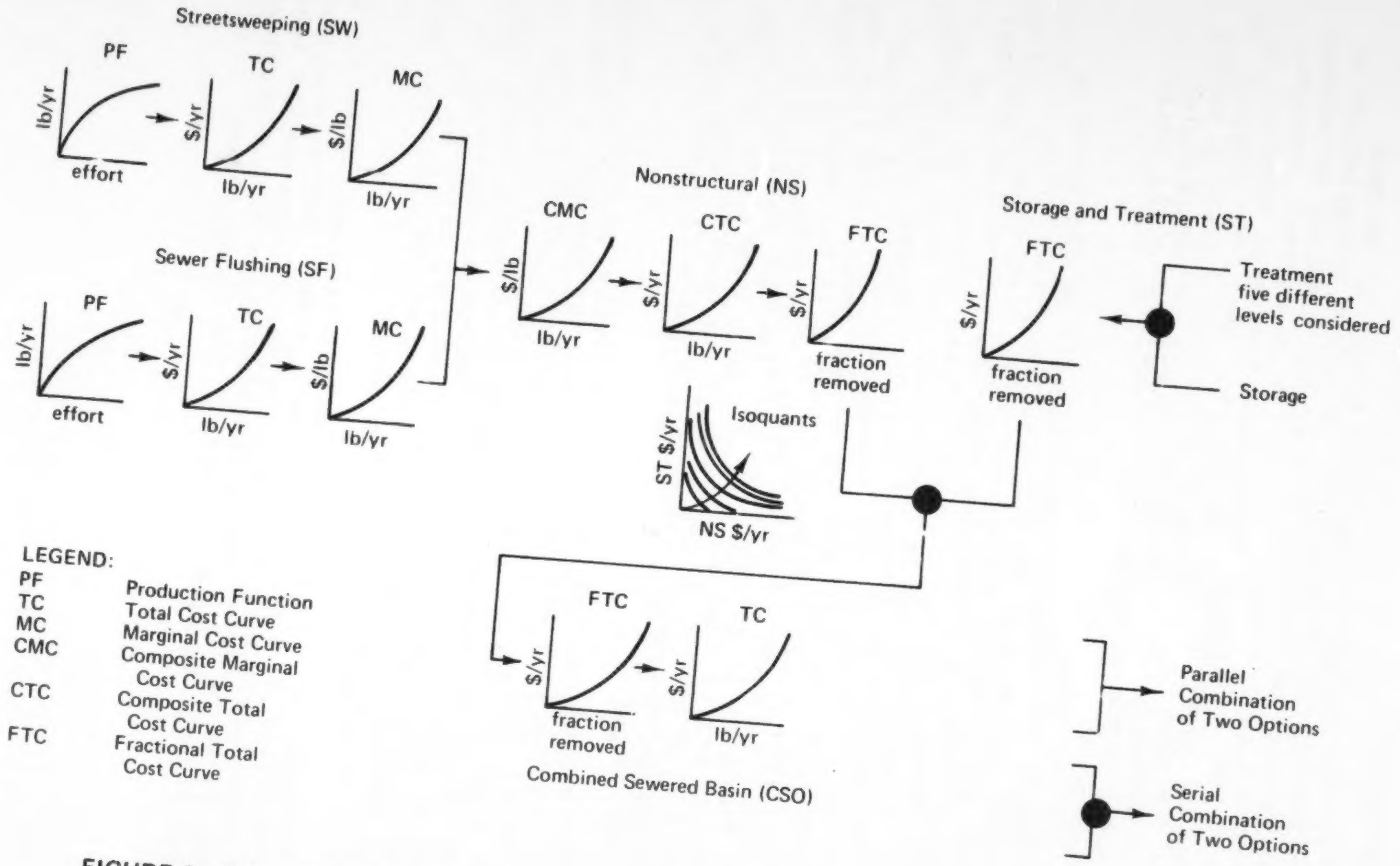


FIGURE 4. Schematic for the economic optimization of control alternatives for combined sewer watersheds.

SW and SF, are combined to produce a nonstructural fractional total cost (FTC) curve, which is combined with the storage/ treatment option, using production theory, to produce an optimum total cost versus pollutant removal curve for the combined sewer watershed.

In order to evaluate point sources, separately sewered (i.e., urban stormwater) areas, and combined sewered areas, total cost curves for each of the above are obtained individually and then combined, as shown on Figure 5. It is noted that the total cost curves shown on Figure 5 are expressed in terms of cost ($/year) versus some measure of water quality improvement (α). The units of V should be in agreement with the water quality goals for the receiving water.

For example, if the goal is to reduce (or eliminate) DO occurrences less than 2.0 mg/l, then α should equal the number of hours per year that DO levels less than 2.0 are reduced. Consider a case where low DO levels (i.e., $<$ 2.0 mg/l) occur for 100 hours per year. Alpha would then be defined on the interval 0 to 100. An α value equal to 75 would represent elimination of low DO levels for 75 hours per year and α equal to 100 would represent total elimination of low DO levels. The transformation between pounds of pollutant removed per year and improved water quality defined in terms of the parameter α is made by applying the results of the water quality modeling. Thus there is a direct linkage between the water quality simulation results, the water quality goals, and the economic optimization.

Once the total composite marginal cost (TCMC) curve for the urban area is established (see Figure 5), then the economically optimum combination of point and nonpoint source pollution controls for any desired level of pollution abatement is known. This procedure determines whether improved wastewater treatment plant (WWTP) effluent quality, urban runoff controls, or combined sewer overflow abatement are desired separately or in combination so that a given water quality goal can be achieved. The degree of pollution abatement required will determine the degree of each control to be used. At this point in the planning process, an economically optimal project or combination of projects has been identified.

PROJECT EVALUATION

Using information obtained from the water quality simulation and economic optimization, it is possible to develop relationships for both optimum cost versus the amount of pollutant removed and optimum cost versus receiving water quality response. Several levels of pollution abatement need to be evaluated by both the water quality simulation and the economic analysis to adequately define these relationships.

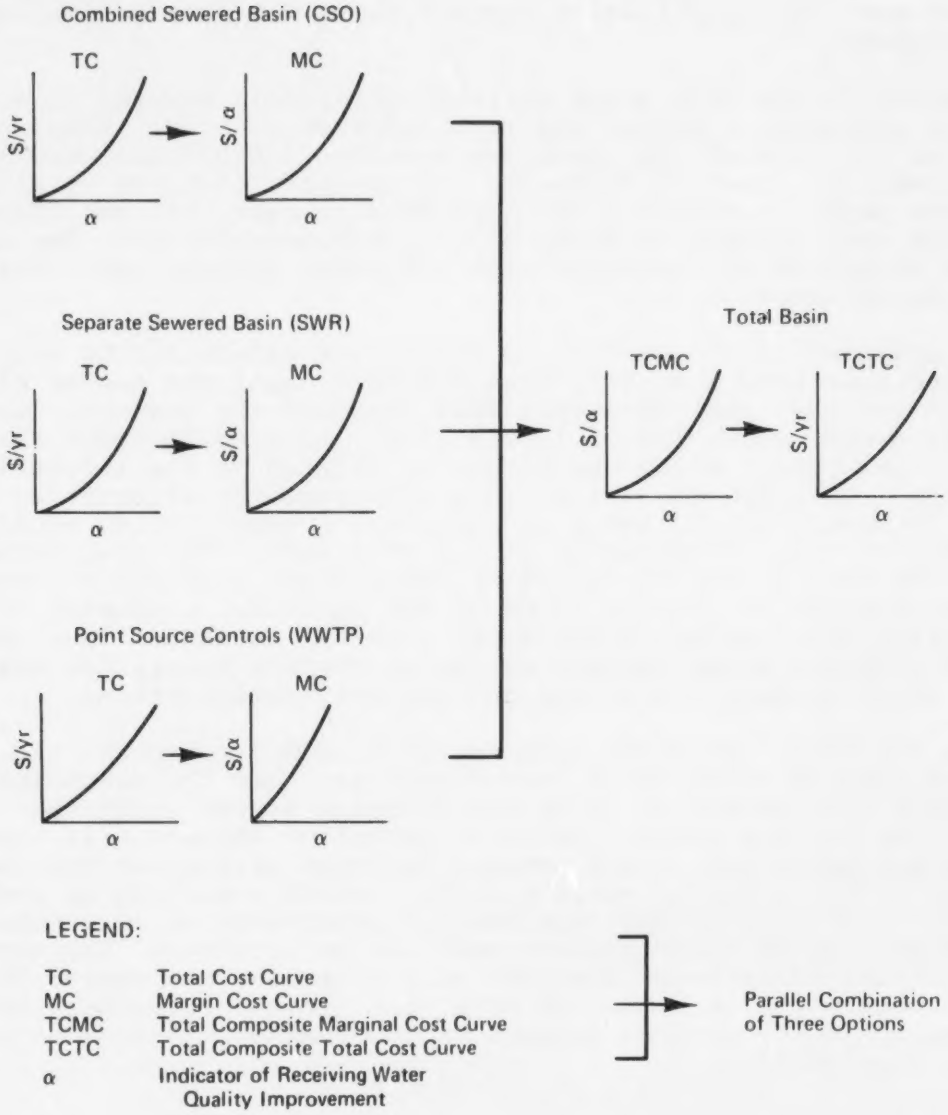

Combined Sewered Basin (CSO)
TC
MC
S/yr
S/α
α
α
Separate Sewered Basin (SWR)
TC
MC
S/yr
S/α
α
α
Point Source Controls (WWTP)
TC
MC
S/yr
S/α
α
α
Total Basin
TCMC
TCTC
S/α
S/yr
α
α
LEGEND:
TC Total Cost Curve
MC Margin Cost Curve
TCMC Total Composite Marginal Cost Curve
TCTC Total Composite Total Cost Curve
α Indicator of Receiving Water Quality Improvement
Parallel Combination of Three Options

FIGURE 5. Schematic for the economic optimization of control alternatives for urban areas served by both combined and separate sewer systems, with point source controls.

The optimum cost versus water quality relationship is an essential key to the project evaluation because it allows decision makers an opportunity to determine whether the costs of achieving a water quality goal are worth the expected improvements in water quality. For example, the hypothetical optimum cost versus water quality curve shown on Figure 6 indicates that the cost of meeting the goal of allowing only one 4-hour period per year with a DO concentration below 2.0 mg/l is four times the cost of allowing five low-DO periods.

Ideally, this question should be addressed in light of expected annual damage resulting from low-DO situations. For example, Figure 7 presents a hypothetical relationship between expected annual damage and annual duration of low DO levels. Damages should be measured in terms of dollars; however, other units, such as number of fish kills, reduction in fish yield, or effects on species diversity, would be useful. Such a relationship may very well show that five low-DO occurrences (Figure 6) are no more damaging than one occurrence. By changing the water quality goal to allow five low DO periods, a significant monetary savings could be realized while still protecting the receiving water's fish and wildlife resource.

Unfortunately, the basic data required to construct a damage versus water quality curve are generally lacking, making this type of relationship difficult to quantify at the present time. Thus project evaluation currently involves a large degree of intuitive judgment, particularly in the area of damage or, conversely, benefit assessment. However, every effort should be made to obtain or develop the basic data necessary to quantify this relationship.

If, on the other hand, the optimum cost of meeting a water quality goal is too high and the goal cannot be modified and still protect the beneficial use of the water, a change in the water's intended use may be required. The cost versus water quality curves could be used to provide guidance to suitable goals and uses that are monetarily feasible. This approach can be used in staged implementation of pollution abatement projects. Consider the case of a stream that once supported a cold-water fishery, but due to oxygen-demanding substances, from wastewater treatment plant effluents, combined sewer overflow, and urban runoff, extended periods of low DO concentrations had rendered the stream unusable even as a warm-water fishery. The optimum cost versus water quality analysis might show that achieving cold-water fishery goals is beyond present fiscal resources, yet a warm-water fishery goal could be achieved. Rather than abandoning the project altogether, facilities required to achieve the warm-water fishery goal could be constructed immediately, with facilities required to meet the cold-water fishery goal to follow if and when funds become available.

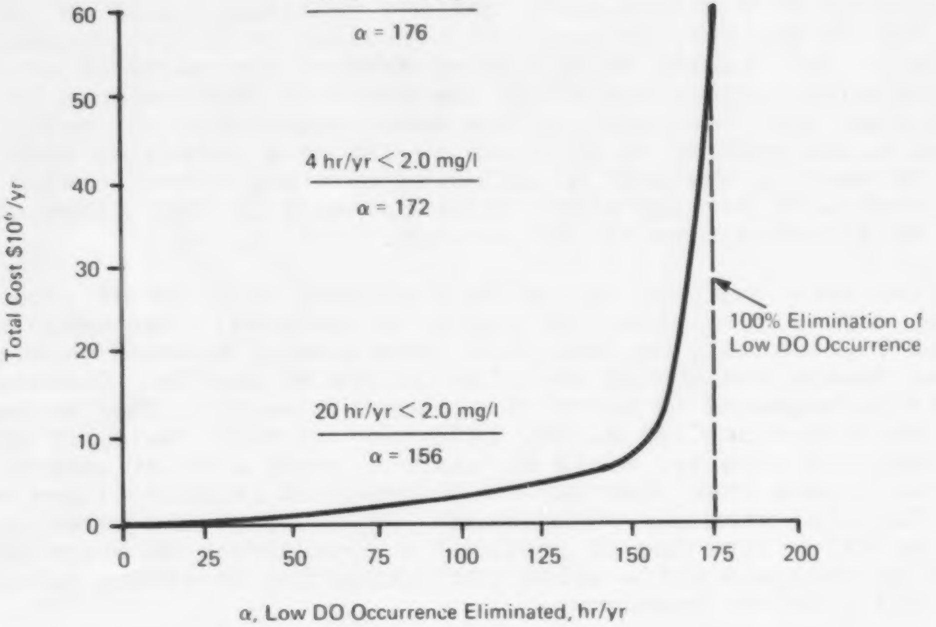


FIGURE 6. Hypothetical cost vs. water quality improvement curve.

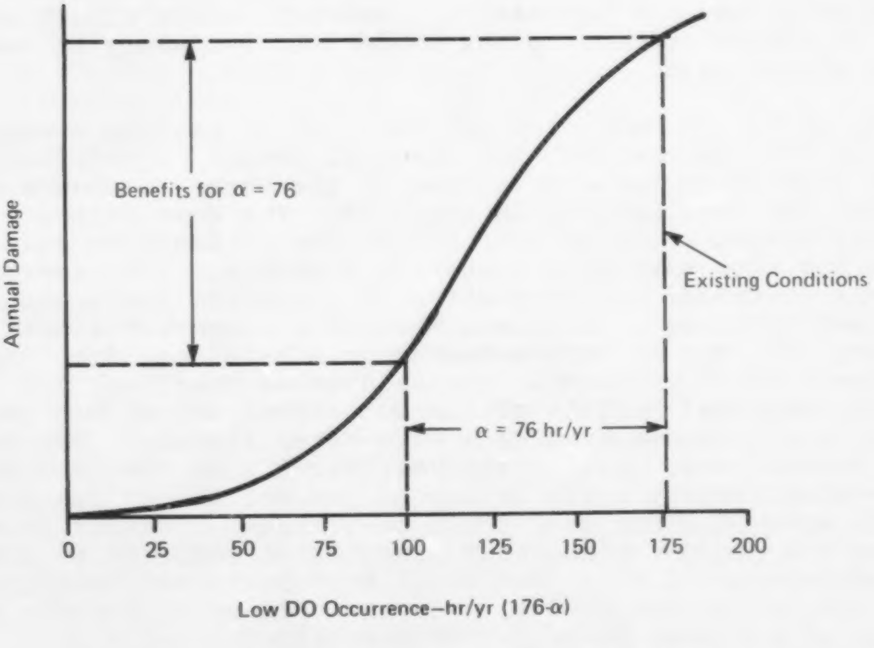


FIGURE 7. Hypothetical damage vs. water quality curve.

PHASE II - DETAILED ANALYSIS

If the results of the Phase I analysis indicate that there are water quality problems that can be corrected with affordable solutions, the detailed analysis of Phase II is required. This phase considers, in much greater detail, the area's hydrology, combined and separate sewer system hydraulics, nonpoint and point source pollutants, and the receiving water quality response. This analysis may require more sophisticated data and models, depending upon the magnitude of the project.

The product of the Phase II analysis is a detailed description of the optimal mix of control alternatives, the total plan costs, and the receiving water quality response due to the plan. The description of selected nonstructural alternatives includes the level of effort required, the area to which the alternative applies, the expected pollutant reduction due to the alternative, and the cost of the alternative. The description of selected structural alternatives includes a preliminary engineering design of the facility, the expected pollutant reduction due to the facility, and the cost of the facility. These are the items generally necessary to complete a Step I (201) facilities plan for the study area.

PHASE I - CASE STUDIES

A brief description of two case studies developed as part of the combined sewer overflow and urban stormwater runoff portion of the 1978 Needs Survey[2] are presented to illustrate selected steps in the water quality planning process. The Philadelphia, Pennsylvania, case study presents the simulated response of the Delaware River to improved WWTP effluent, and to combined sewer overflow and urban stormwater runoff controls. The Des Moines, Iowa, case study demonstrates the linkage between water quality simulation and the economic optimization procedure.

Philadelphia, Pennsylvania

The City of Philadelphia is located in eastern Pennsylvania at the confluence of the Delaware and Schuylkill Rivers. The study area consists of the City of Philadelphia on the western bank of the Delaware River and Pennsauken, Camden, and the neighboring urban areas on the eastern bank of the river. This area is heavily industrialized and contains many industrial waste sources as well as numerous municipal wastewater treatment plants. Approximately 45 percent of the study area is served by combined sewers, while the remainder is either drained naturally or is served by storm sewers.

Continuous observed DO data for the Delaware Estuary at Chester, Pennsylvania, are available and were used to calibrate the receiving water response portion of CSPSS, as shown on Figure 8a. This figure shows good agreement between simulated and observed water quality, particularly in the range of 2.0 to 6.0 mg/l, and further indicates that the Delaware Estuary is subject to low DO (i.e., less than 2.0 mg/l) levels approximately 40 percent of the time.

The results of simulations with different pollution abatement alternatives are presented in Figure 8b. Based on the simulation, improved treatment of domestic and industrial wastewater can result in significant improvements in the receiving water quality, more so than storage and treatment of CSO and urban runoff. However, even with very high levels of municipal wastewater treatment, DO occurrences at or below 2.0 mg/l cannot be completely eliminated.

A cost analysis was not undertaken for Philadelphia, but it was considered in the next case study.

Des Moines, Iowa

The Des Moines urban area is located in central Iowa at the confluence of the Des Moines and Raccoon Rivers. The study area consisted of the Des Moines urban area tributary to the Des Moines and Raccoon Rivers. The extent of the combined sewer watershed is about 8% by area. The remainder is served by separate storm sewers or by natural drainage. The receiving water consists of the Des Moines River below the confluence with the Raccoon River, extending to Red Rock impoundment approximately 45 miles downstream. Major BOD sources include upstream flow (81%), municipal wastewater effluent (8%), combined sewer overflow (1%), and urban stormwater runoff (10%).

Results of the simulation indicate that under present conditions, minimum DO levels in the Des Moines River are less than 2.0 mg/l 134 hours per year and less than 5.0 mg/l 1,480 hours per year, about 17% of the time. These results indicate the existance of a water quality problem by the criteria developed previously. Furthermore, a typical stream quality "standard" would call for minimum DO values to be greater than 5.0 mg/l "at all times." This standard would be violated 17% of the time under existing conditions.

The simulation also indicates that a high level of control (i.e., BAT) for all controllable BOD sources, including WWTP effluent, combined sewer overflow and urban stormwater runoff, will eliminate all occurrences less than 2.0 mg/l, but DO levels less than 5.0 mg/l cannot be eliminated. Even with BAT of all controllable BOD sources, simulation results indicate that minimum DO levels will be less than 5.0 mg/l, approximately 900 hours per year, or 10% of the time. Thus, a typical stream standard requiring that DO levels always be greater than 5.0 mg/l

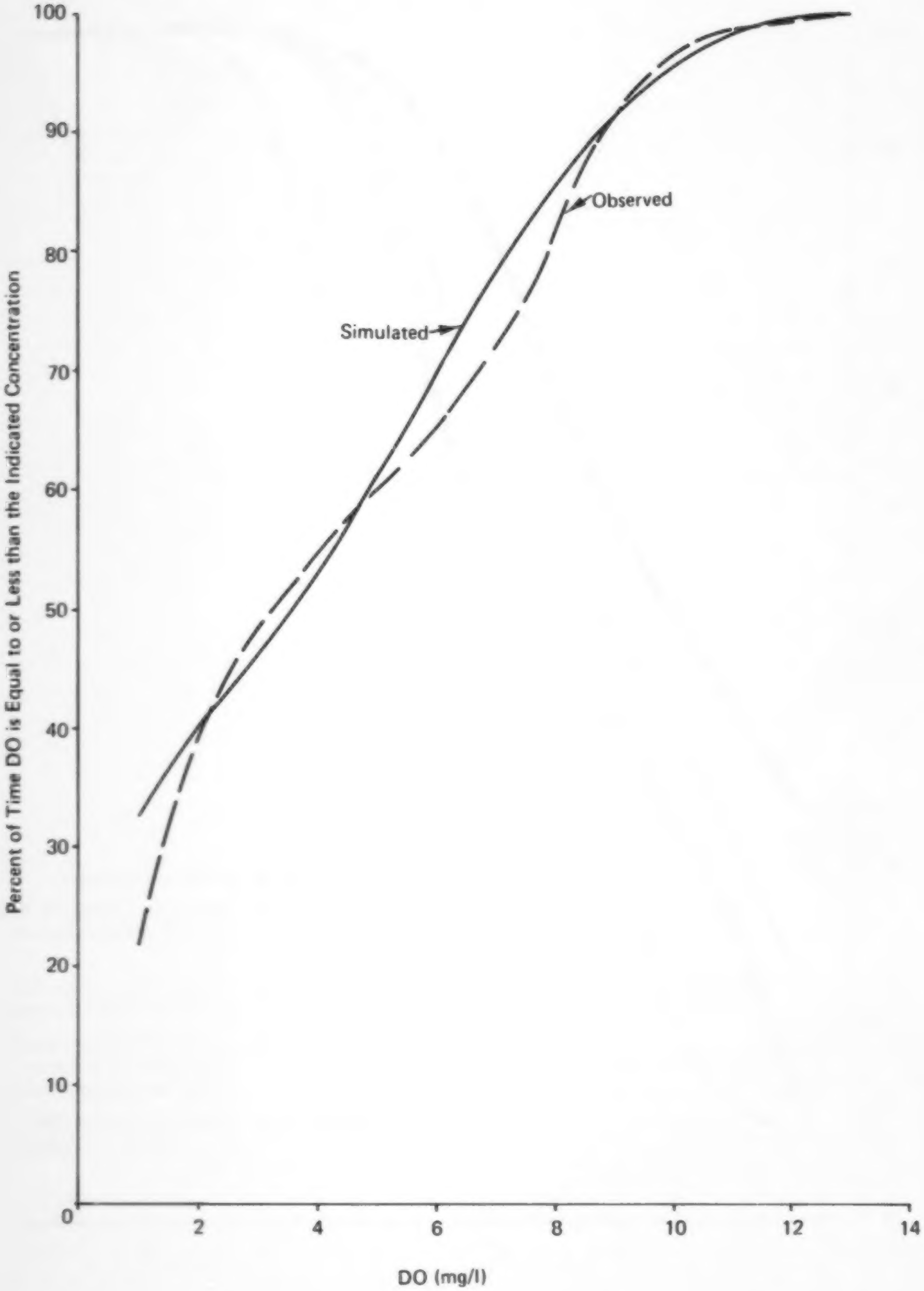


FIGURE 8a. Receiving water dissolved oxygen calibration at Chester, Pennsylvania.

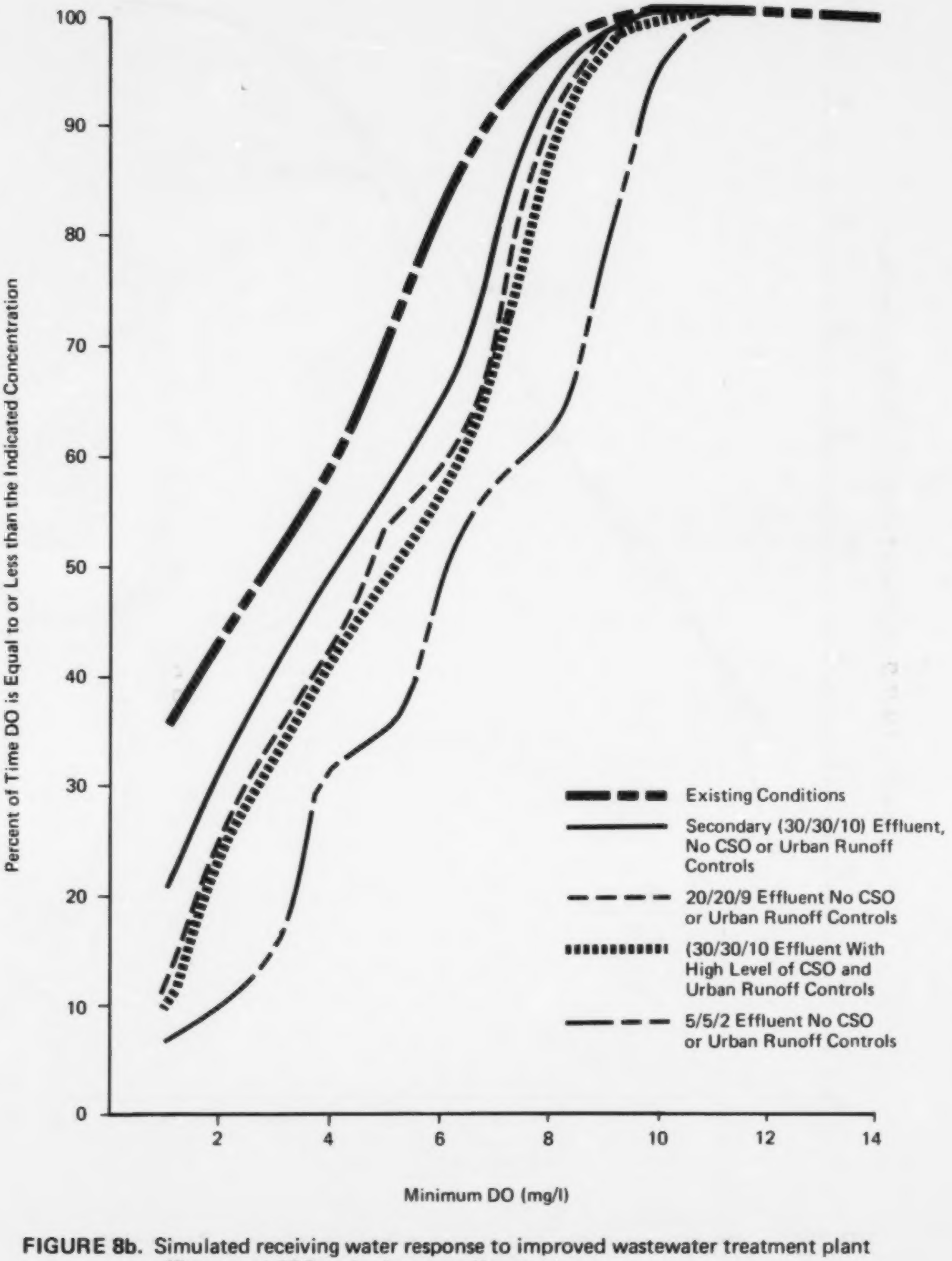


FIGURE 8b. Simulated receiving water response to improved wastewater treatment plant effluent and CSO and urban runoff control.

cannot be met regardless of the degree of pollution control provided. Therefore, the economic optimization of control alternatives presented here is based on minimizing the occurrence of DO levels less than 2.0 mg/l.

Figure 9 presents production functions for point and nonpoint source control alternatives developed from multiple 5-year computer simulation runs. The output, or indicator of performance α, is defined as the number of hours per year DO levels less than 2.0 mg/l are reduced and the level of effort is defined as the amount of BOD removed, in lb/yr. These production functions indicate that additional WWTP (i.e., point source) controls are largely ineffective in reducing low (<2.0 mg/l) DO levels, whereas urban stormwater runoff (SWR) and CSO controls are generally more effective.

This conclusion is valid for this study site, but these results cannot be generalized to other areas since a case-by-case analysis is required.

Figure 10 presents estimated total cost curves for point and nonpoint source control alternatives. The wastewater treatment cost curve is the estimated equivalent annual cost of pollutant removal beyond the secondary level and the CSO and SWR cost curve is the equivalent annual cost of pollutant removal for the optimum mix of CSO and SWR controls.[2]

Marginal cost analysis reveals that for the entire range of water quality improvement considered, CSO and SWR controls are more cost effective (i.e., lower marginal cost) than additional point source control. The optimum areawide total composite marginal cost (TCMC) curve and the total composite total cost (TCTC) curve are presented on Figure 11. These curves provide valuable information to the decisionmaker relative to the economic tradeoffs of water quality improvements.

The marginal cost curve (Figure 11a) illustrates the increasing extra costs incurred when total or near total elimination of low DO levels is the water quality goal and the total cost curve (Figure 11b) illustrates the increasing total project costs. For example, in order to eliminate all DO levels less than 2.0 mg/l, we must be willing to pay up to $350,000 for the last low DO hour eliminated and $6.2 million per year for the total project. Alternately, if we decide that DO levels less than 2.0 mg/l are acceptable for a period not to exceed 4 hours per year (i.e. $\alpha = 130$), then we have also decided that we are willing to pay up to $270,000 for the last DO hour eliminated and $5.2 million per year for the total project.

One provision of PRM 75-34 states that the marginal cost of control should not be substantial compared to the marginal benefits. For this case, if α is considered an adequate measure

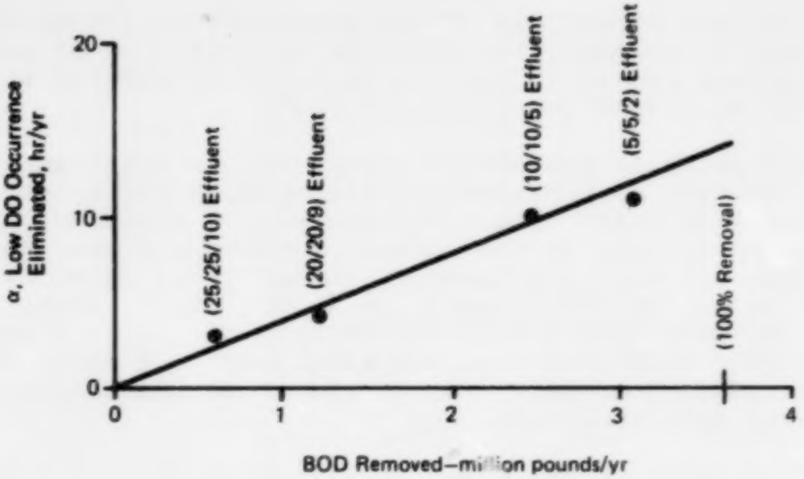


FIGURE 9a. Production function for wastewater treatment beyond secondary, Des Moines, Iowa.

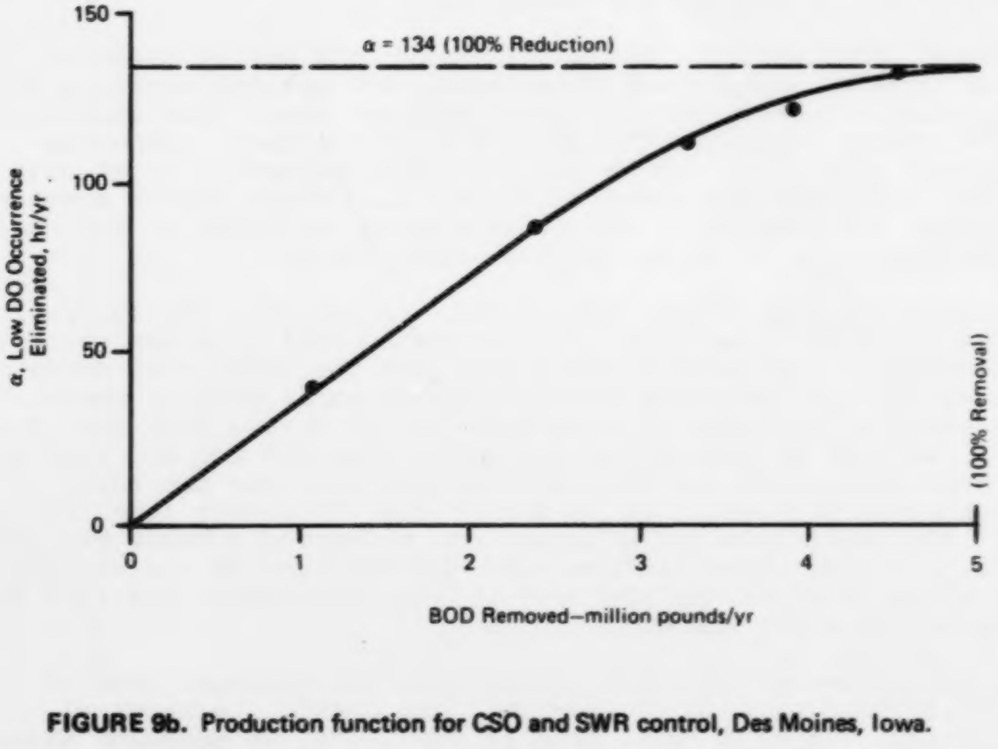


FIGURE 9b. Production function for CSO and SWR control, Des Moines, Iowa.

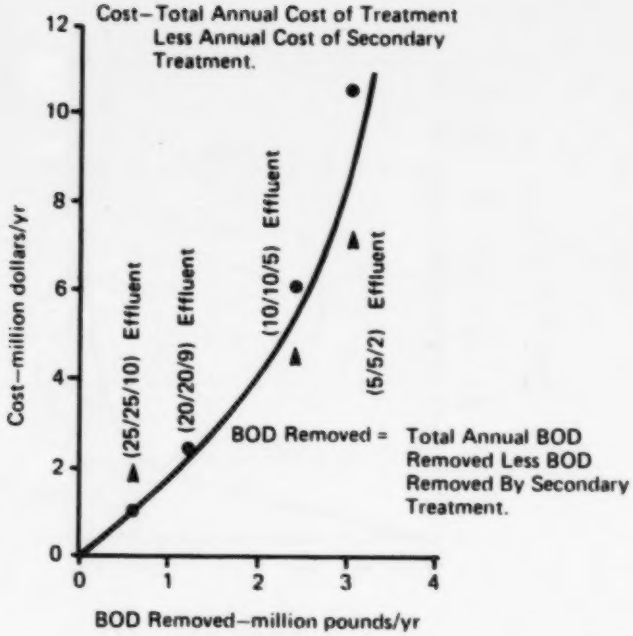


FIGURE 10a. Estimated cost of wastewater treatment beyond secondary for 40-mgd plant, Des Moines, Iowa.

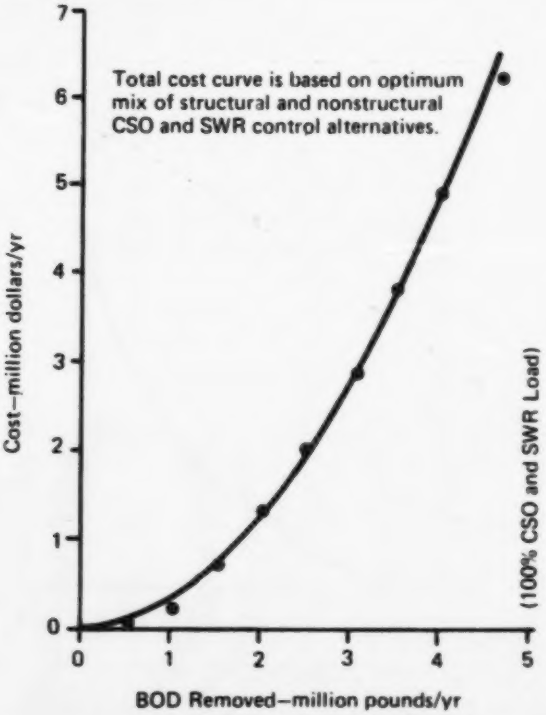


FIGURE 10b. Estimated cost for optimum CSO and SWR control alternatives for Des Moines, Iowa.

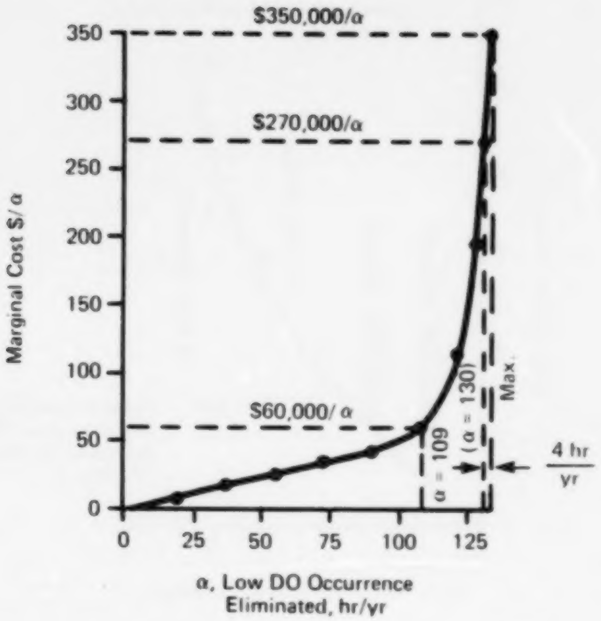


FIGURE 11a. Marginal cost curve for optimum control alternatives, Des Moines, Iowa.

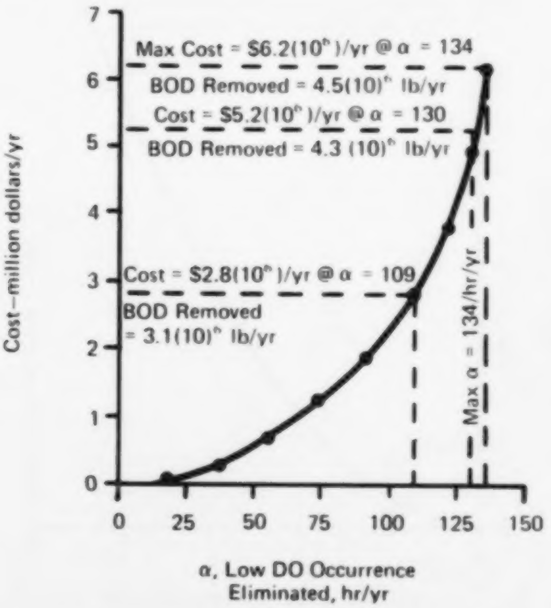


FIGURE 11b. Total cost curve for optimum control alternatives, Des Moines, Iowa.

of benefits, the implied water quality goal is elimination of 109 of the 134 hours per year with receiving water DO levels less than 2.0 mg/l. Marginal cost in this case would be $60,000 for the last low DO hour eliminated and total cost would be $2.8 million per year.

It is obvious that a decision relative to acceptable receiving water quality will have a significant impact on total committment of both Federal and local tax revenue. It is also obvious that a relationship between the water quality indicator and actual environmental damage (see Figure 7) would be of great value to the decisionmaker.

These case studies illustrate the site-specific nature of the water quality planning process. Continuous water quality simulation of the Delaware River below Philadelphia indicates water quality improvements due to AWT and little improvement due to SWR controls. On the other hand, continuous water quality simulation of the Des Moines River below Des Moines indicates nominal improvement due to AWT compared to CSO and SWR controls. Considering the number of physical, biological, and economic variables involved, it is unlikely that the exact same mix of controls or receiving water quality goals will provide the optimal solution for more than one situation.

SUMMARY AND CONCLUSIONS

Water quality planning in an urban area is a complex and often interative process. Once the goal of secondary treatment of municipal wastewater is met, identification of residual water quality problems and their cost-effective solution must be addressed on a case-by-case basis. This paper has presented a general framework for water quality and wastewater facilities planning which consists of answering seven basic questions relative to the urban area and its receiving water. These are:

1. What is the intended beneficial use of the receiving water?

2. What water quality goals or criteria are required to ensure this use?

3. Based on these goals, is there a water quality problem?

4. Can combined sewer overflow, urban runoff, or additional point source pollution abatement techniques solve the problem, and what degree of control is required?

5. If the problem can be solved, which controls, in general, are the most cost-effective?

6. What is the nature of the tradeoffs between cost, degree of control, and receiving water quality?

7. Considering these tradeoffs, are changes in the water quality goals or in the desired beneficial use of the receiving water indicated?

These questions can be answered by application of tools currently available to the water quality planner, including (1) continuous hydrologic/water quality simulation and (2) economic optimization of control alternatives.

Urban water quality problems arise from intermittent nonpoint sources of pollution, such as combined sewer overflow and urban stormwater runoff, as well as from point sources, such as municipal and industrial wastewater effluents. The intermittent and extremely variable nature of nonpoint source pollution requires consideration of receiving stream interactions on a continuous basis. Selection of "design events" for analysis will not give the water quality or facilities planner a true picture of the behavior of the receiving water or of the impact of proposed pollution control facilities.

Economic optimization procedures, such as production theory and marginal cost analysis, are available and are applicable to the urban water quality planning process. Considering the availability of practical economic optimization tools, there is no reason why suboptimal solutions should be considered. It is always possible for a given set of economic circumstances to determine the optimum mix of feasible technologies which will achieve a given level of pollutant removal from one or more sources. In many cases, it is also possible to link the results of continuous receiving water quality simulation to the economic optimization and thus establish the optimum mix of control alternatives required to achieve a given water quality goal or an array of alternative water quality goals. Given these economic/water quality tradeoffs, the most appropriate level of water quality improvement and/or project cost can be chosen. Knowledge of actual environmental damage or benefits versus water quality indicators would be invaluable in this decision-making process. Unfortunately, such information is generally not available.

ACKNOWLEDGMENTS

Credits

The development of continuous stormwater pollution simulation system (CSPSS) and the case studies were funded as part of the 1978 Needs Survey for control of pollution from combined sewer overflow (Category V) and urban stormwater runoff (Category VI), contract No. 68-01-3993. Philip H. Graham, P.E., was the project officer. The authors would like to thank their colleagues at CH2M HILL for their extensive support and encouragement in the preparation of this paper.

REFERENCES

1. U.S. Environmental Protection Agency. "1978 Needs Survey--Cost Estimates for Construction of Publicly Owned Wastewater Treatment Facilities." EPA 430/9-79-001 (FRD-1). (February 10, 1979).

2. Wycoff, R. L., J. E. Scholl, and S. Kissoon. "1978 Needs Survey--Cost Methodology for Control of Combined Sewer Overflow and Stormwater Discharge." EPA 430/9-79-003 (FRD-3). (February 10, 1979).

3. U.S. Environmental Protection Agency. "Quality Criteria for Water." (July 1976).

4. Gehm, H. W., and J. I. Bregman (ed.). Handbook of Water Resources and Pollution Control. Van Nostrand Reinhold Company. (1976).

5. Shepard, M. P., "Resistance and Tolerance of Young Speckled Trout (Salvelinus Fontinalis) to Oxygen Lack, with Special Reference to Low Oxygen Acclimation," Journal of the Fisheries Research Board of Canada 12, 387-446. (1955).

6. Wycoff, R. L., and M. J.Mara, "1978 Needs Survey--Continuous Stormwater Pollution Simulation System--Users Manual." EPA 430/9-79-004. (FRD-4). (February 10, 1979).

7. Heaney, J. P. and S. J. Nix. "Stormwater Management Model: Level I--Comparative Evaluation of Storage-Treatment and Other Management Practices." EPA 600/2-77-083. (April 1977).

10

WORKSHOP ON PRACTICAL APPLICATIONS AND RESEARCH NEEDS FOR RECEIVING WATER RESPONSES TO URBAN STORMWATER

SESSION I

Moderators: Francis J. Condon, EPA
James P. Heaney, University of Florida

Speakers:
1) James P. Heaney, University of Florida
2) Raymond P. Canale, University of Michigan
3) Thomas L. Meinholz, EcolSciences, Inc.
4) Robert E. Pitt, Woodward-Clyde Consultants
5) Miguel Medina, Jr., Duke University

Mr. Condon

Five minute presentations will be given by Messrs. Heaney, Canale, Meinholz, Pitt and Medina to summarize what they said yesterday and give you time to put down some questions. After they all have finished then we'll open it up for discussion. You may have questions related to special project findings, problems encountered and associated solutions, receiving water impact deficiency and research needs, short term and long term effects, and assessment of the adequacy and reliability of the data base for receiving water studies. Can you identify the most crucial gaps in deficiencies in the data base? What is the role of the indicator parameters? How can we decipher urban wet weather impacts from dry weather impacts? Is it worth having a complex model or would we be better off with more of the black box models? The last word I would like to leave with you, the price you have to pay for this discussion is to identify for us some of the real world research needs, which will then also give us justification for trying to get the funds to accomplish them. So it works two ways.

1) Dr. Heaney

Good morning. I think the message is pretty clear, we've only just begun to look at receiving water impacts and I would like to suggest we start fresh. The Sutron work in particular, looking at the continuous dissolved oxygen over a year, two years, three years, suggests a lot of different ideas that have been traditionally used for looking at the water quality standards. So, I think that's one definite area. Associated with that is the whole question of reliability, and we found numerous cases of incidences of spills, and we've got the seven day, 10 year minimum flow still in the back of our mind. What about the reliability of sewage treatment plants? How frequently do they spill? Industrial waste discharges? How well are your ordinances being enforced against discharges? Would you really know of some of these discharges if it wasn't that bad? All of these are going to give you a system reliability that's going to determine what your limiting ability is to control pollutants. We need to look at the percentage of the time we can expect to achieve certain quality water conditions. We've also been doing work on a data base for EPA for four or five years trying to collect what information is available. It's very costly and expensive to do that. As far as research needs on that, I don't see anything too glamourous that needs to be done except there is a lot of hard work in checking and verifying the data. I think there has been kind of a loss of "back to basics". There is not good quality assurance in data collection, reporting and dissemination and we found even the best data still has a lot of garbage in it.

We thought it was going to be an easy project - we get the data on a tape and then you just put that on a larger tape and then make that available to users. But, if you really get in there and look at that data, there is a lot of bad information in there and we don't have good checking procedures. We need to establish some quality assurance in data handling that we don't have now. There are a lot of ways for processing data retrieval storage systems. The capability is out there, but I still think we have to do a better job of putting it in clean so it will come out clean.

As far as modeling goes, during the last four or five years, you couldn't mention the work "model" and hope to get any money for research. Instead, you have to talk about model application. Maybe we have over-modeled, but there is still a need there to do more fundamental work. Maybe we have enough of the differential equations models. We don't have standard methods yet for models, we're just beginning to think about this, and I think this is a fruitful area for some research.

The other problem we dealt with is the question of economic efficiency, cost effectiveness, and you have heard a lot about that, pro and con. There is a lot we can do selectively to find those receiving waters whose beneficial uses are most seriously impaired. Economic optimization should be based on selective control of the most serious problems and beneficial uses of receiving water. When you go to selective abatement, the burden of proof lies on the regulator to prove that, in fact, this is the best way to go. So the analysis part of the whole pollution control problem becomes a much larger cost.

2) Dr. Canale

I would like to share some problems that I have encountered in the application of receiving water quality models for evaluating the impact of CSO's. I want to talk about development, calibration, verification of the model and some problems encountered in trying to abuse these calibrated models for water quality projections.

It is possible to develop a standard type differential equation mass balance model to simulate the event of a combined sewer overflow problem.

The response of Onondaga Lake to a storm in September, 1976 shows an increase followed by the die-off of the organisms. Also, a similar kind of response was noticed for oxygen in various parts of the lake. These responses can be modeled but we need various kinds of physical data, more biology, circulation, dispersion, material loadings from point sources, agricultural sources, urban runoff, etc., reaction rates, and then finally we need to input various environmental conditions such as temperature for the case of dissolved oxygen. Many of the models are very sensitive to the activities of algae, so we need to know something about light, light extinction, incident light, the level of algae, chlorophyll and nutrients. These have to be specified prior to making calculations, so during the calibration or verification stage, you may be able to measure these factors and simply use them as input parameters in your model.

The problem comes up, what do you do for, let's say, chlorophyll-a level when you are doing a projection? You have this model, you've calibrated it, you want to make a projection, you need to assume something regarding the chlorophyll, nutrients, pre-storm conditions, light and temperature. Well, if you are in a situation where the assumption you make regarding the chlorophyll level can be an overwhelming factor and be even more important than how you may manipulate material loadings. In that case, the problem turns out to be pretty arbitrary depending on what you assume for chlorophyll.

During this period, June to September, 1976 in Onondaga Lake, you can see that the surface dissolved oxygen was varied from ten milligrams per liter all the way down to less than 2 milligrams per liter. This is an overwhelming factor that overshadows the impact of any combined sewer overflow. At the same time, the chlorophyll is doing the same kind of variation. These kinds of variations, independent of the combined sewer overflow, are very serious problems in applying models. If we are dealing with different parameters for control, a different time scale for the different kinds of modeling approaches may be required.

In this particular case in Onondaga Lake, we went through and developed a model for chlorophyll and BOD and TKN and DO which were daily average values over a season which could then be input and used as initial conditions or baseline conditions for any model attempting to respond to a short time event such as a CSO. We also would recognize that this time scale problem would be an important consideration in other kinds of non-point source control, evaluation, and of course point source control evaluation as well.

The Genesee River in New York showed the critical DO sag as a function of the volume of discharge of CSO for various levels of treatment. This was with zero treatment, and a small amount of discharge to increase the amount of discharge the critical DO increases. This whole graph may be sliding up and down by several milligrams per liter depending on what was assumed for the incident level. This is a very important consideration because you can see that dependent on what you assume, you may be above or below some standard.

The final point would be that in order to evaluate the cost benefit type of considerations in this kind of problem you must have more capability than simply being able to project or calculate an incremental effect of the combined sewer overflow or urban runoff. You must have the ability to project the overall value or overall concentration.

3) Mr. Meinholz

Some of the unique problems that occurred in Milwaukee related to sediment. We were not able to identify in our early stages of the project, the source of the mechanism that was causing the drastic DO problems in the Milwaukee River. Through long term field investigations we were able to identify that sediments were there and causing the problem; that it was more of a physical problem, a scouring action directly related to overflow volumes and rainfall intensities.

In identifying sediment sources in a municipal area such as Milwaukee, where the downstream is nothing more than Lake Michigan, we tried to quantify the upstream sources, the sediments in the storm sewer areas and what they were contributing, as well as the combined sewer areas. What about instream velocities? Are they critical enough to scour sediments and how does it impact other receiving streams that are not like Milwaukee? How does the depletion exert itself as a function of sediment scour?

The long term buildup capabilities of different areas, different streams are also another research need that we have identified. The Milwaukee sewer district is undertaking a program to identify what would happen if sediments are removed. If you have to remove sediments and buildup will reoccur again with time, it is awfully difficult to justify the expenditure in order to do something like that.

What are the effects of various control strategies? If Milwaukee goes to deep tunnels or any municipality goes to sewer separation or some type of satellite treatment can we, in fact, through a limited program since construction is quite a few years away in most instances, can we verify some of our predictions of what the improvement in water quality will be? If in fact, we can predict the improvement, can we then relate it to benefits? That is probably the biggest need right now.

The whole area of toxics we have touched on in a number of topics in the last two days. In Milwaukee we are trying to relate this to sediment problems. Years and years ago when the industries, in and around Milwaukee, especially the plating and canning industries, were discharging to the rivers, some of the sediment core samples showed evidence of very high levels of cadmium and other metals two to three feet below the water-sediment interface. These toxics are directly related to some of those industries in the lower reaches and we will be paying for these mistakes that were made by industrial as well as municipal discharges in years to come. If sediment scouring or dredging occurs, do all these toxics come up to the surface? The impact there may become much greater than if we simply went with abatement techniques without dredging.

In Milwaukee the difference between wet and dry weather impacts is extremely noticeable. In other municipalities it may not be as dramatic. We identified our wet weather impact strictly on a physical basis.

And finally the whole portion of complex versus simple models. Complex models were sold as being the answer, yet in the final analysis we're still some way from the final answer. It's the basic understanding of the problem and the ability to predict the basic response to the problem that's most important. It is required to identify in a simple model what the problem is, what the source is, and what the mechanism is. After we have identified the source and mechanism using simple models it may become possible to use the complex model for more verification.

4) Mr. Pitt

Basically, most of the research effort that has to be expended in the next several years should be directed to eliminate beneficial use degradation. Right now I think we really have to examine site specific biological degradation and relate that to the chemical conditions in the receiving water. A major biological concern would be relating the loss of the habitat where the reduction of a component in the aquatic food chain would reduce the number and type of a specific organism in the habitat which is the ultimate component of concern. If we are concerned about fish, that doesn't necessarily mean that we monitor fish, but critical components for the food chain. Now, there are many reasons why they also monitor fish, but fish are easy to catch and relatively easy to identify, but because they are mobile they are very difficult to relate to a specific location in the receiving water. If your receiving water is not homogeneous, which is most likely the case, a lot of their food materials are somewhat less mobile and therefore somewhat easier to relate to specific problems within their receiving water. The beneficial use impairment should be based on short and long term conditions. Short term is a lot easier and that is basically what we have always been concerned with historically, relating to water quality conditions.

We have also found that sediment conditions can exert longer term biological degradation in receiving water and we don't really have a very good handle on how to address that as far as the need for control. You need to monitor sampling locations affected by various levels of degradation. Control test areas and intermediate situations should be monitored in order to try to find or establish an acceptable biological degradation situation and then relate that to the chemical conditions found at that location and affecting that from upstream areas. You have to sample similar habitats or a family of habitats at all your stations. You have to be careful about locating specific sampling points near local points of interference such as areas where there is an excessive amount of bank erosion, or ground water infiltration. You have to worry about locating extremely close to something like an outfall where the specific effects from one outfall might overly influence smoother gradation of conditions that you are attempting to look at and of course the micro analysis of the degradation from a single outfall as we heard from Seattle is an extremely important consideration also. But, you have to separate the physically short-term versus the longer term types of problems that you are observing. The biological sample, has to include representatives of various levels of the food chain including invertebrates, algae, fish, etc. As I mentioned, fish are relatively easy to sample and identify, but quite difficult to relate to a specific chemical condition at a location.

5) Dr. Medina

My comments are basically directed to the fundamental research needs and problems from the complex interactions and interrelationships of urban stormwater controlled systems and receiving water responses. When we look at the large number of variables I think they can be grouped broadly into three main categories: a natural science perspective in order to get a better understanding of the chemical, biological and physical phenomena; we can look at them from an engineering perspective to achieve some type of useful purpose, for example deterministic or statistical prediction of transfer processes, storage treatment, the design of conveyance and containment systems and so forth; we could also look at it from a management planning perspective to achieve some degree of control over the state of the system within the context of a rational decision-making process. The basic engineering problem has always been to obtain amicable relationships between the inflow and the outflow processes, and the variables describing the state of the system.

For example, such variables as density, volume, temperature, and so forth can be considered state variables. Decision variables can be how much storage and treatment you are going to implement. Regardless of whether you group these variables in any one of these categories, we have to converge. We have to use a multi-disciplinary approach and converge towards a unified purpose. I describe that as acceptable levels of receiving water quality. Among the generally recognized needs, I'm going to quickly go through some of these; geophysical, hydrodynamic characterization of natural receiving water systems.

We still don't know enough about the geophysical hydrodynamic problems. Especially, identification of the transition. We have fresh water, tidal rivers and estuarine systems. What happens in the transition and what physical-chemical biological reactions are unique to these? Specification of heterogeneous kinetic interactions of all the sequentially reacting substances - we don't know enough about that. We could model better if we had more basic research in this area, such as boundary effects on various substances at the air-water interface.

We also need to establish better analytical relationships for land eroded suspended matter, and I think that one of the real problems is how to get a good handle on the up-land erosion. I think once we have the sediment in a particular channel, we can transport it, but relationships like the universal soil loss equation are too highly empirical and they are not storm event dependent. In other words, there is a lot to be done in that area. A fundamental understanding of the accumulation and final disposition of synthetic pollutants produced by our chemical and industrial firms through the natural food chains that exist in the wastewater treatment systems, as well as in the receiving water quality systems, is needed.

We need to have a better handle on the dynamics and control of wastewater treatment plants and stormwater storage treatment systems. We need a better integrated receiving water response to time-variable point and distributed waste inputs. We also need to continue to develop refined methodologies within an analytical framework that will evaluate cost effectiveness for these urban wastewater control strategies in terms of ultimate receiving water quality benefits.

Those are general comments. Now I want to sort of get into some specifics; problems that I feel are critical in receiving water quality modeling for urban stormwater management planning.

Getting back to kinetics formulation, we have eight or nine equations for reaeration rate for streams, but we need to have more fundamental research on the separate and combined effects of surface active agents at the air-water interface, reduction of turbulent mixing by high suspended sediment loads and dissolved chemicals. All of these tend to reduce the oxygen transfer rate. Some work by Covar, for example, is of some interest whereby he presented a general approach to selection of the appropriate reaeration equation for streams, now a similar approach is needed for lakes and estuaries and hasn't been done. There really hasn't been enough done on the relative magnitude or importance of all the parameters in these equations either.

If we get into the deoxygenation rate, the k_1 factor, as we like to call it, we still use a first order rate model, which does not include specific growth rate which makes it really a second order reaction. I think more basic research is needed there. The whole traditional BOD model is under challenge. We also do not have a good handle on the time dependency of all these rates during high river runoff conditions. Very little work has been done in this area.

Now, going to assessment tools, two basic assessment tools you need are instrumentation and mathematical models of some sort. I'm not as qualified to talk about instrumentation as somebody like Phil Shelley who is in the audience and has done quite a lot of work in this area, but everybody knows we need better instrumentation. I think one of the critical needs is to optimally design the networks, both the water quantity, rain gauge and stream flow, as well as the water quality networks using a better methodology. In other words, analyzing the character of the hydrologic signals and then determining the response time that is required for the instruments and so forth.

Going into mathematical models, I will only briefly say that we need to simplify first and then use the complex models. General guidelines for model selection, I think, is the key issue. Most of you do feel like you need some general guidelines for selecting your particular model for a particular situation. Now model development needs again span such things as simplification of routines for storage treatment, for dual purpose operation control systems, and complex mechanisms including transport. I want to get down to the planning and management techniques and very briefly say that in a full scale testing of all of these management techniques, such as detention, storage and so forth, we have to quantify benefits and unless we relate water quality pollution strategies to the benefits derived, we are really going to fall on our faces.

Moderator - Dr. Heaney

Would anyone like to make a comment? Ask a question? Besides Peter Moffa. I'm going to make a few comments, and I hope they will be responded to. I have to agree that one of the areas of the toxic picture, and a very important part at that, is to evaluate the toxics in terms of the specific chemistry of that receiving water and the synergistic effects. But, what I really want to get to, and it relates somewhat to the toxics, is that we've got to begin now to collect the data. You know we can do stormwater sampling and wet weather impact work as part of the CSO element of facilities 201 planning. This is fundable, it's been accepted, somewhat hesitantly. You can go out and sample before a storm, during and after a storm, but you can't spend very much money collecting that background data. So, I think, we have got to work through our EPA representatives here as well as back home in the regional offices to get them to recognize that we've got to establish a funding vehicle to begin to collect this data now to go buy a data base and not wait until you go in with a CSO or proposed CSO program. I think it's all related to what you've said and I'd like to see some responses.

Question: Is there anybody who knows EPA allocations for data collection?

Answer: I haven't been able to trace down the figures, but the percentage of the total EPA dollar that goes into the hardware is quite high and the amount that goes into the data collection of studies has definitely gone down over the past years because of this need to move ahead and implement more construction grants. So, hopefully, we're seeing a shift a little bit away from that, and a little bit more towards analysis and forethought before you jump into these things.

Mr. Greg Welter, O'Brien & Gere Engineers, Washington, D.C.

I have a comment regarding the question of biological benefits. We have had some people that have approached this by doing natural biological monitoring longitudinally down the stream and they have been able to establish a variation as we go further downstream. But, as far as a practitioner or somebody working for a client and trying to recommend facilities and demonstrate a retrievable benefit, we haven't seen those studies carried to the point of being able to say, "Well, if we implement this alternative we can see this kind of change, we'll see this difference."

Where we've got a degraded habitat or something like that, we'll see a different biological regime. Now, in terms of the analytical receiving stream models where it's attempted to project those kind of differences, we've had to take one step back in terms of going to the chemistry of it. For instance, going to dissolved oxygen when your real benefit that you are looking for is fish. I don't really care whether it's fish or mayflys or whatever, it appears to me that we are able to do reasonable dissolved oxygen models, that we know a fair amount about reaeration and things like this, but in the absolute case we've got red book values that tell us what the criteria are for fish toxicity in terms of low dissolved oxygen. When you actually get out to a particular case, in this particular river, I can demonstrate that there aren't fish there that used to be there before man was there. People do like to fish so there is a benefit. I can demonstrate that there is periodic dissolved oxygen contraventions so that's a possible cause, but I defy you ever to find somebody that will definitely say that dissolved oxygen, or for that matter, anything else, is the cause. You will never find a biologist that will say, "This is the reason that we don't have the fish here."

Do biologists want to come up with a definitive statement? I think you'd say the eastern United States are experienced in trying to get data. People are just used to no fish.

If you look at most of the developed urban areas in our lifetime, it has always been that way. So, in these areas the Illinois Historical Society is doing a very worthwhile thing. You have to go back to the turn of the century to when the receiving waters were still in fairly good shape and they have got some data on the fish populations at that point that can show the long term degradation, the loss of fish, and then the recovery. But, if you are in a developed area and you talk about, "Gee, we're going to make this swimmable and fishable," most people will laugh at you because they teach their children at a very young age not to go near those waters and they accept that. So, there is a whole education process developed there. Now, in the western United States in the newer areas, there is much more sensitivity because you have a very good hope of anti-degradation, but in the east and a lot of the older areas this is not the case. We found there are no fish kills in many of these areas because the fish are long gone. So, I don't know what you conclude from that.

Dr. Al Zanoni, Marquette University, Milwaukee, WI

I'm not a biologist, just a form of request. We at Marquette University are starting on the project where we are looking at some of the sludges and residue that will be generated from storm treatment devices, physical, biological and chemical characteristics, so if anyone is aware of any device anywhere in the United States or elsewhere, that might be available for treating stormwater, we'd be very happy to hear about that and possibly get some material for the study we are doing.

Mr. Mike Lopez, U.S. Geological Survey, Brandon, FL

We are starting a program in the USGS, Tampa, and one of our concerns is the high cost of data collection. Practically over half the cost will go into analysis of the samples. Now these prediction models are planning models using the data that we put in and we estimate the pollution loads which we will try and verify by collecting data. The amount of data collected would affect the certainty with which we would be able to estimate model proficiency.

A modeling conference sponsored by EPA was held in West Point about six months ago to discuss various model applications, model accuracy and the confidence people have in models. It is very easy to make small changes in model coefficients and find large changes in model projections. If the level of change in the model projection is large compared to potential changes we might get from a management alternative, then we are in trouble in order to be able to justify such things. We may need to develop a family of criteria to show response due to different concentrations and duration. Also, in the case of a river model with let's say, fifty segments, what generally is done is to pick the lowest of the fifty and try to meet the standard in that one segment. We should realize that only one segment out of fifty is in a strain and that we are trying to protect fourty-nine segments that are doing alright and maybe the fish are avoiding the problem.

Dr. Marty Wanielista, University of Central Florida, Orlando, FL

We've had some experience on the selection of the storage areas within a watershed and essentially it is an economic trade-off that you're dealing with, especially if you have the treatment capacity at the end of the pipes. The major problem that we ran into was this: if you get into a downtown urban area in which you have a limited right-of-way, the municipality has a responsibility of either getting an easement or fitting something into a right-of-way, and there is where your problem develops. It is fitting something into a right-of-way because most of the streets, if you look under them as in downtown Orlando, there is a six foot box that's filled with electricians wiring and to relocate some of these would be very expensive. The comment is that it is difficult to find the space.

Moderator - Dr. Heaney

We may have frequency curves depending on the average time of hourly data, weekly data, and monthly data, but we do not have a response curve to calculate our benefits. We certainly don't have it for a series of events. If you have data indicating 20 percent kill of the fish, what happens to the remaining 80 percent? Are they unstressed, partially stressed? If you talk to biologists they just laugh at you. I don't think that method is going to take us too far, because I think you just have to go back case by case again. For example, the Rhode Island Lab of EPA has spent a couple million dollars already trying to do comparisons on when you close beaches. They got some data from somewhere around Egypt where there are grossly polluted beaches and people are swimming in them, versus trying to discern for the U.S. beach. They go out and talk to the bather and say, "Are you a swimmer? How do you swim? Did you actually stick your head under water? How do you feel?" It's very expensive to collect information, but they are doing these studies. I think they said many are working in coastal zones now and they are going to move to fresh water areas. It's very expensive but there are some numbers that are beginning to come.

Mr. Dick Bain, Brown & Caldwell, Seattle, WA

I wanted to make a non-definitive statement about qualification of benefits. I wanted to go back to the days before anti-degradation, which was 10 or 12 years ago, and mention that at least in the Federal agencies at that time there was a challenge that many of us had to develop dollar-benefits for either benefits or declarements to Federal type projects. These were co-engineering declarement-type projects but we were dealing with water quality effects and we had to work closely with agencies like the Bureau of Outdoor Recreation and the Sports Fisheries and so forth. I was working on the west coast with what was then the Federal Water Quality Administration, and we did develop curves which would go to Cincinatti and pay homage to the water quality biologist there. We tried through knowledge of what the fishery resource was, what the sport fishing and the commercial fishing experiences were, and the like to come to some sort of systematic approach for arriving at a dollar benefit or detriment to a particular project. We did some lakes, we did some rivers, and we did some estuaries in the Bay area of San Francisco. There was an attempt working with things that existed like Senate Documents and other things like that and there is some relationship between quality and the resource. In this way we tried to come up with dollar figures and they were used. I wish there were some better way to get these "would you believe curves" so that people would believe them.

Moderator - Dr. Heaney

I think your comments are certainly borne out by our literature search. There is a gap. There is some very good work that was done in the 50's and 60's and you will see that in literature and then it kind of dies out in the last ten years. This is because it is no longer part of the game.

Ms. Madeline Snow, Mass. Dept. of Env. Quality Engr., Brighton, MA

We have two NURP projects which are going on, and our needs are for very practical information. Short of sponsoring 10 or 12 graduate students to do detailed biological studies in our streams, I need information about how to get a handle on the impact of stormwater on the biological community. We are interested in very simple information, if it is at all possible to get. Second, I would like to emphasize to any of the researchers here today, do not underestimate the social, political constraints out there in the real world. The models may be beautiful, the curves may fit, and the projects will still not fly. So, just be aware of those constraints because they are very real and they are the deciding factor.

Mr. Mike Terstriep, Illinois Water Survey, Urbana, IL

I have some mixed feelings and I'll try to put some of them together. I've seen some of these earlier meetings where we were all hydrologists and we were talking about the urban problems and so forth. Then we had chemists come aboard and now we have biologists aboard. I think we finally may have the right group of people to talk about these problems, and we certainly have to learn to communicate. I was impressed with Dr. Lee's work and some of his comments concern me. I think he said something to the effect "I'm wasting my time to put together a table of chemical values and doing more quality work." I can certainly see after this meeting that this is true to a certain extent. However, I have been working on storm sewers so there is not much of anything to count down there, so I'm going to stick with water chemistry.

I also think that Dr. Lee has to soften his view a little bit and look at chemical constituents when he is doing his bioassay and his hazard assessments, and he has to learn to relate to those chemical constituents. Arbitrary standards are here to stay, I think they are a reality. There are too many regulatory agencies and we have to have something that they can get their hands on. Something so they can go to the lab and get quick answers. We've seen laboratory instruments expanded and become fantastic tools that will give us these larger array of constituents. I think that biologists must also relate to these things that are in the real world. I would like to give you one little story from a lady in Massachusetts on the public and the fact that they are becoming more sophisticated. The other day, we have a NURP project going on also, I was installing a rain gauge. School was out on that day and there was a nine year old boy who came over to me and asked what I was doing. I showed him the rain gauge and said we were putting these rain gauges out. He said, "Well that only tells you that it rains right here in this one spot. What about on the other side of the street over there?" I thought, "My God! I know professional hydrologists who haven't come to that conclusion yet." So, we have some pretty tough customers coming up in the future.

Dr. Jim Falco, U.S. EPA, Athens, GA

I work primarily in rural areas rather than urban, but I thought I would share some of the experiences we have had, particularly with relationship to these frequency-duration calculations and the statistics that might be involved in more realistic standards or more quality goals. In relation to the evaluation of pesticides which is our major concern in the rural area, we do, in fact, use frequency duration curves in the Office of Pesticide Programs that estimate the impact of used pesticides in receiving water bodies. There are some limitations that we have found.

Basically, if you work in the areas where there is not a research project where there is only USGS or core data, the highest frequency that we can get is essentially daily. We can't go to four hours or eight hours and I suspect that in urban areas you run into the same problem. In relation to another comment that was made in referring to the statistical approach mentioned, the statistics that we used started with DiToro's work and added in the duration parameter. Our limited experience in checking the density of the rain gauges and so forth to adequately characterize an area is that you can do fairly extensive areas even when you are concerned with the summer-intense small thunderstorms. We did some checking between 10,000 square miles and essentially one rain gauge every 1,000 square miles and found that the pollutographs that we generated showed very little difference, which is encouraging. However, I would temper that with saying that, that is one basin in the southeast, so experiences in other countries may be different. We found that the major determinant in variation in area dealt more with the distribution of land uses than variations in rainfall pattern.

The last comment I have regards sediment scour. That of course is a major problem in the areas we work in. There are a number of models which are either just completed or will be completed in the next six months and it might be beneficial for people in the urban area who are concerned about scouring and movement of material after deposition in previous time to search that literature for information.

Mr. Skip Ellis, CH2M Hill, Reston, VA

In my presentation yesterday, I mentioned economic optimization. I have one sort of problem in that the methodology we use comes out with a global optimum. This is the cheapest way, overall, to solve our problem. But it is not necessarily the cheapest way for a municipality to solve their problem because we may be using some control measures that are not funded by EPA - street sweeping, management controls, things of that nature - and, if we are going to make a push towards using economic optimum solutions, I think some thought has to be given, or maybe it has been given at EPA, to funding items that have not been funded in the past. Otherwise, if I was a municipality director I would say, "That's fine but you're going to bankrupt me, so let's put in the storage treatment because I can get a 75:25 split and I can afford to do something."

We have a report coming out that addresses that question specifically. The dilemma for the Federal Government, as I see it, is that they recognize that it is a problem but they have no mechanism for funding O & M. They cannot make a long term commitment for O & M. They can make commitments for construction grants. So, I think there is a recognition of the problem, but that may be a topic for other discussion.

Mr. Robert Molzahn, Harza Engineering Company, Chicago, IL

A comment was made about 30 billion dollars of the Needs Survey earmarked for coliform control and in a lot of the big cities we have a lot of storage in people's basements for combined sewer overflow and if it is a public health problem on beaches, is it then also a public health problem in the basements? What is EPA's position on this? I can respond to that one because I have cleaned out my basement in Skokie, Illinois five times a year for four years. Government accounting offices just looked at the public health benefit. In fact, I may be involved in some of the studies. They went around and surveyed people. "Did anyone get sick after you cleaned out the basement of all of the combined sewage in it?" They couldn't find anyone that really got sick from that. They thought maybe someone got a skin infection or something. Chicago is going through a whole reassessment trying to quantify benefits. But, this is one that they are looking into. It's kind of a unique problem.

Mr. Dennis Athayde, U.S. EPA, Washington, D.C.

From my perspective on controlling urban runoff and combined sewer overflows, wastewater treatment plants has been the easy way out for the initial cleanup. We've done the easy part and the hard part is remaining. Control of combined sewer overflow is probably where the next emphasis is going to be on. We talked to the construction grants people. They talk about having 26 billion dollars earmarked for combined sewer overflow control. Is it a federal problem or is it a local problem? We are taking the position that it is a federal problem and we are putting it in the construction grants program. The money is going to be available for combined sewer overflow control so there is no real question. But, that's the answer to your question. The money is going to be there.

We made the decision that we are going to put money into the combined sewer overflows. The next decision that we have to make is, are we going to put money into separate storm sewer discharge cleanup? The act was changed and the funding for the treatment of storm sewer discharges was not allowed any more for a five year period. We need information on whether or not separate storm sewer discharges are a problem. What we need is a good definition of what our problem is, and Madeline tried to point that out. We have to be able to describe what our problem is. We have to have something to hang our hats on.

Mr. Steve Tindale, City of Tampa - DPW, Tampa, FL

We've made a decision that the analysis should be case and site specific. We've run around in circles in terms of how to analyze problems and we've said we have to go to the sites and look at them. Maybe, when we talk about beneficial use we need to talk about site specifics. Maybe we need to get away from thinking about some of these general indicators. We need to go to the local public and ask them what the beneficial use is. Maybe the federal role is looking at major streams, national impact, major recreational areas, or something of major impact on major communities, not small communities. Maybe what we're talking about is trying to come up with a process that says "Let the people decide the options, all the benefits, and the cost and come up with the beneficial use."

When we put the burden on the local government, maybe what we can talk about from the beneficial use is some kind of procedure to be developed by the EPA, that states, etc., so the local people can take a local problem and describe beneficial uses and the cost that they are willing to pay to obtain those beneficial uses.

Dr. Fred Lee, Colorado State University, Fort Collins, CO

I feel very strongly on that approach and in the City of Fort Collins, where we're working now, we are developing procedures to specifically answer that kind of question, how you take chemistry and biology information and put that together to assess what the public will get from the expenditure of so much money. You lay it out to the public and say, "This is what you'll get."

. . . Coffee Break followed by continuation of the workshop . . .

SESSION II

Moderators: John L. Manicini, Manhattan College
Richard Field, EPA

Speakers:
1) Richard D. Tomlinson, Municipality of Metro Seattle
2) Raul S. McQuivey, The Sutron Corporation
3) John L. Mancini, Manhattan College
4) Martin P. Wanielista, University of Central Florida
5) Peter E. Moffa, Stearns and Wheler

Mr. Field

Good morning - I think the earlier workshop session went very well. I got a lot out of it. There were a lot of interesting remarks made. I'm sure these comments, many times, were just general and vague but they can start to make you think. Now, if anyone has any comments we are going to have more time after the second set of speakers. We are going to make an attempt in the formal proceedings to have this workshop included, but I say it is an attempt because these things could be a problem. I don't know how the tape recorder is working and things like that.

We were talking about having Phil Shelley make some comments and I noticed before the coffee break we missed him because all of a sudden the discussion picked up again. I would like to give Phil that opportunity now. I'm sure he'll have something important to say.

Dr. Philip Shelley, EG&G Washington Analytical Services, Inc., Washington D.C.

Some of us have expectations of R & D that perhaps aren't really appropriate. To me, it is not the function of the research dollar to solve immediate problems. To solve today's problems today. Rather, the research dollar should be used to anticipate tomorrows problems and to try to come up with the combined bag of tricks that we need to effectively deal with those problems. Now we have concerns for certain things and emphasis shift to another set of concerns. The pendulum swings back and forth and I think it will always be that way. The R & D community has responsibility to maintain the technology base so that the proper perspective can be applied as best we can at all times. What are our real R & D needs as I perceive them - problem identification, we've talked a lot about that today but so much work needs to be done there and although problems are locally specific - the problem exists in the eyes of the beholder.

There are some things that are a little more universal that we can address so that people in the public participation process can be given the opportunity of choice. The R & D program owes the public alternatives and explanations of these alternatives in clear enough terms so that they can be understood and the people can, in fact, choose. We talk about the technologies that are involved, we talk more about the economics that are involved because the notion of affordability is really beginning to come home. We do not have an unlimited supply of bucks to pour into any given public concern arena, environmental or otherwise.

If we look at the AWT program and we see some of the localities that now have a beautiful plant and say, "Oh my God, I don't know how to run it - I can't afford to run it, I'm running it ineffectively and I don't know what to do - you know, maybe we need to come up with better designs that can be run by the mayor's son-in-law." Like Madeline indicated, there are institutional problems that we have to take into account. It is inappropriate to expect EPA to fund research efforts that don't in some fashion meaningfully relate to its essential mission as a regulatory agency. We talked to some extent about the need for standards, criteria and research that will support regulations and legislation.

When I look at this horrible thing called water quality, it's more than we can say grace over. Also, we sometimes tend to over apply some of our particular pet little route skills that we know a whole lot about, and we lose sight of the bigger picture - we overly complicate because we know what to do. I look at modeling of stormwater as it started 10 years ago, dominated by the plumbers, because if there was one thing we know how to do it was to route water around. We understood the hydraulics of this situation so the models were very good in hydraulics. They were lousy in water chemistry because we didn't know much about it and besides the people working on the models weren't by and large chemists. When we combine the physical-chemical characteristics, overlay these with the ecological considerations, that the little critters we are concerned about, we have a very complex system indeed. We do, however, have a responsibility to look at new tools as they come along and to continue to step back and to try to maintain, regain or establish fresh insights. DiToro's talk yesterday was an example, one case in point, Miguel Medina mentioned common filtering, another very strong tool that we have got to apply in water quality work.

Time series analysis and life cycle cost analysis are other areas where we have a great application. We focus on the capital cost portion and sometimes you give lip service to O & M and you say "OK that will be about 10% of my invested capital" and you wing it at them. And I submit, it is very poor, if you are not examining the manpower requirements, the training requirements of those people, you may find that the nation's labor pool is unable to support some of the solutions that the R & D comes up with.

Finally, I think that we can develop uniform methodology despite the local specific nature of the problems. We can use the same approaches. It doesn't mean that the answers, as I grind through this uniform methodology, would be the same for Kansas City as they would be for Washington D.C. or Los Angeles, but the approach can be the same.

Lastly, when we talk about benefits - we've been talking a lot about benefits because our awareness is being opened more and more in the area of Proposition 13 - about who benefits. Because we have to determine not only who benefits but who pays, and that is an area not for the technician but for the politician. But, insofar as we can establish no way that mankind benefits from the presence of these particular critters in this given creek, it also doesn't mean that we know everything. We need to continually maintain a bit of a humble posture because we may learn something next year through a well-applied R & D dollar that will point out to us why critter X did happen to be important and he has a way sometimes, if he vanishes, of making us pay in ways we can't even foresee now.

1) Mr. Tomlinson

I'm not a modeler but I do have some very deep interest in the discussions that have been transpired this morning. We do have the situation where we have existing very visible fisheries, noteably our salmon fisheries, and the people in the Northwest, specifically Seattle, are very much plugged into the mystique of this whole thing. It is their way of life and it's very, very important to them. Because of this, aquatic scientists in the Northwest have even stronger urging by their public to meet these questions in a practical manner and attempt to relate to the fisheries. I myself, personally feel kind of sorry for the bugs, the crawlers in the sediment, too; but I have to face the fact, that people in general do not relate well to bugs in sediments. What they do relate well to are things that they encounter in their fishing experiences and their other water recreational activities. So that hopefully, if we can approach our responsibilities by attempting to preserve fisheries, the bugs and the sediment will go along for the ride. Now, with this sort of thing in mind - I guess one thing more that I should mention - is that one of the things that we have done in the Seattle area, this is specifically in metropolitan Seattle, or METRO the organization that I represent, is the setup that we call the salmon enhancement program. We have taken three streams that are partially degraded at this point but are still capable of sustaining salmon fry eggs going through development to fry, and then hopefully, will support salmon fisheries with the idea that we plant salmon there now and allow these fry to go to the ocean to return in approximately four years time. We have four years in which to clean up those streams·and make them a viable habitat for those returning salmon. Now chances are, that if we don't do a good job, the returning salmon may be unsuccessful in their spawning and we will see a very gross failure of our efforts. Either way, it is a kind of a no-lose situation. People in that area are very critically interested in this sort of thing. If they clean up the streams and they have a success to look at, they are going to be very proud of that. If the streams are not cleaned up - or for one reason or another not able to produce a success - that is going to be very visible. That's something that you can point out to the public by saying, "Look, those streams are not fit for those fish that you put so much credence in. It won't support them."

So, here is the emphasis - for us to do the job as far as showing benefits through our research work. I will say, now, the information I am about to present here, in my way of thinking it does not represent a state of the art effort. What I am going to talk about is a brief assessment of water quality impacts on the ecology in the fresh water environment and this plugs into the talk that I gave on Monday. The reason that it does not represent state of the art is because I consider that right now for this type of assessment, state of the art probably gets into multi-varied analysis and that sort of thing, and this is a much more simplified approach; but beyond that, thinking of Dr. Lee's comments, this does not even begin to really think in practical terms about what it means to fisheries. And the reason that we didn't get into that, quite frankly, is that we didn't know how to do it. Dr. Lee said that he has some techniques that are becoming very successful in this direction and I'm certainly looking forward to talking to him about those in some detail.

Our general approach in a very simplified manner of presentation was to try to determine the relationships between numbers of organisms with distance from the outfall using regression analysis. Assumptions being made, of course, since you don't have the outfall at a control site and you don't have a distance from outfall effect. You consider that zero, so anything significant in terms of distance from outfall effects shows an impact from that outfall. Now we essentially summarized our data in three ways and I'm going to show a couple of brief slides here, and I'll discuss them briefly.

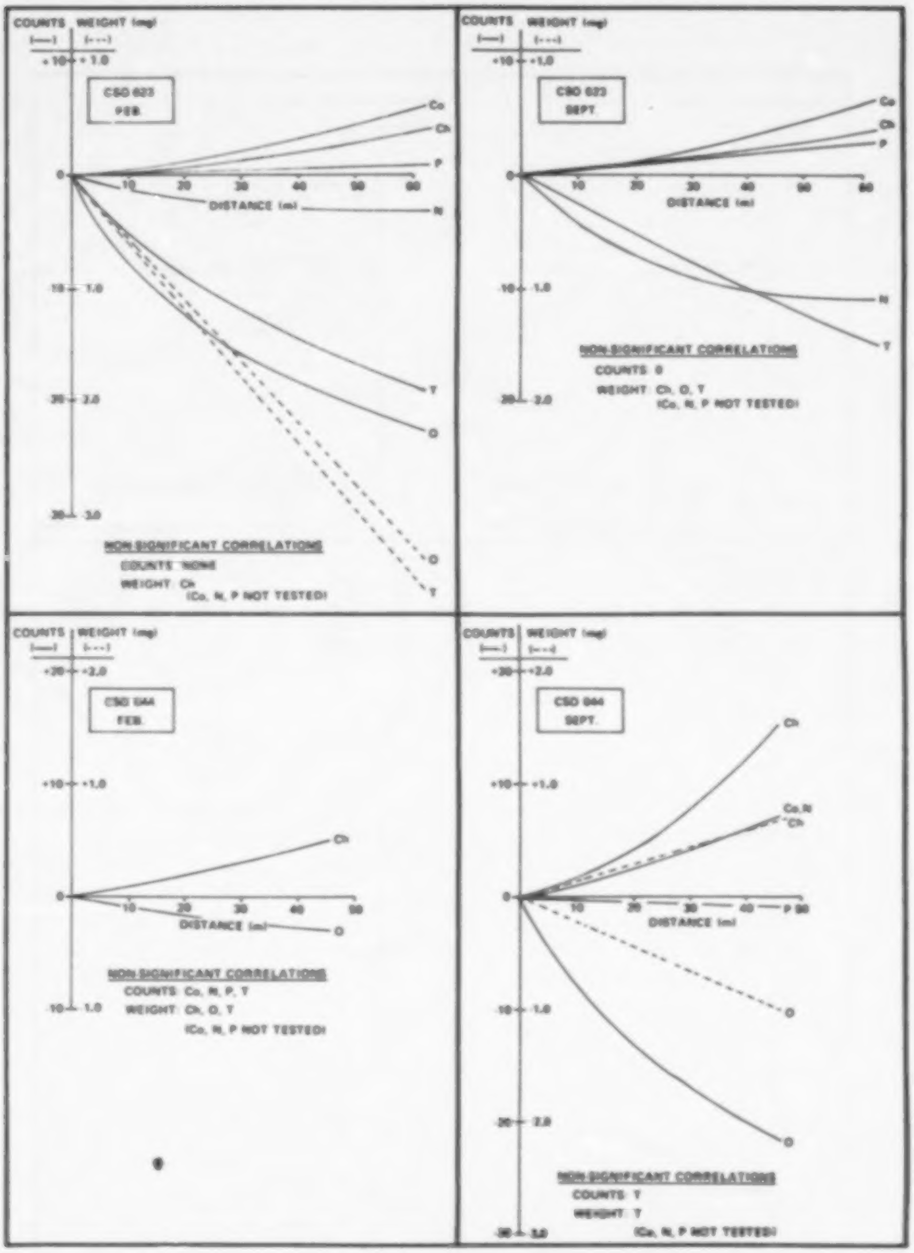


Figure 1. Net change in numbers of organisms and biomass per core, as a function of distance from the combined sewer and storm drain outfalls in February and September, 1978. All relationships determined by regression analysis and significant (at the 95% confidence level) unless otherwise noted. Ch: chironomids, O: oligochaetes, Co: copepods, N: nematodes, P: pelecypods, (Pisidium spp), T: total for all organisms.

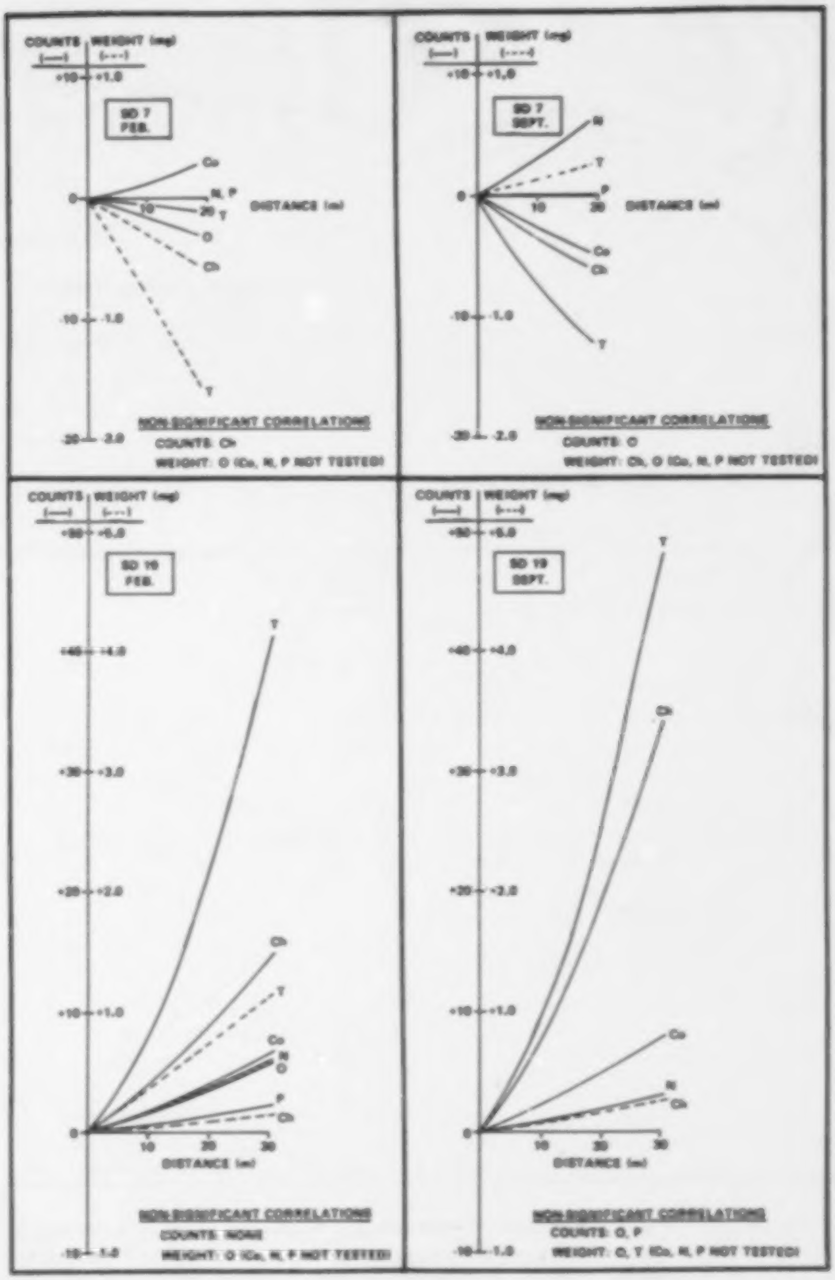


Figure 1. (Continued)

We also calculate coefficient of determination, which is the square of the regression correlation coefficient. If I could have the first slide, please. OK, what I'm showing here are on the top from left to right is represented a single outfall, in the bottom left to right a second outfall. They are both combined sewers. On the left hand side of the figure we have one season, February, which is typically part of the rainy season where we have a lot of overflows and on the right side we have the drier season. I'm showing here relationships of the changed number of organisms and the change in biomass per core in relative units, with distance from the outfall.

There was an enhancement as you move toward the outfall of oligochaetes and this is for a polluted type condition and it has been mentioned several times during this conference - that's a fair indicator. So, move on to the next slide please. For the storm drains we ran into some extra problems and basically these consisted of two things: it is difficult to get a proper super imposition or sampling grids, and I think we missed a significant part of the effluent plume. Secondly, we saw in these conditions significant scouring right by the outfall which depleted the number of organisms immediately next to the outfall. This had a tendency to strengthen the positive correlations and weaken the negative correlations, so we had plots like this and we were not able to use a straightforward interpretation. Indeed, inspection of the data showed that this occurred for oligochaetes only because of the scouring next to the outfall. In other words, what I'm trying to say is, from the storm drains we saw enhancement of oligochaetes from the outfall.

To encourage discussion a bit further I will mention that we did make a general attempt to get a ball park figure of what the depletion or enhancement of these organisms might mean in terms of fish tissue. Now, this involved a number of assumptions which are very gross, but it was a matter of essentially trying to get an idea of what the order of magnitude might be. Taking the most extreme condition, where we had found that enhancement of the oligochaetes, which in this case were responsible for about or better than 90% of the number of organisms in the sediment of that outfall. Essentially, what we did was, first determine the area of impact by relating what we saw at this outfall to what we saw at control sites at similar depth and then using that area as a basis we integrated the numbers of organisms that we saw increased within this area which we were able to calculate using a couple of other assumptions. One was a general figure of dry weight to fresh weight conversion for oligochaetes that we pulled out of another paper and the other was to use a 10% conversion factor for the amount of fish tissue that you could expect to be produced by 100% of your given value. We came up with some interesting figures that might give a ball park figure as to what these impacts might be on fish. This depends on the assumption that all of these organisms are available for consumption and the fish are successful in finding all of them. That would mean, in terms of fish tissue, the biomass that we calculated for the oligochaetes total enrichment was something like 47.3 kilograms around this single outfall and this outfall represents a substantial CSO, one of our more frequently overflowing CSO's in the Seattle area, and that translates to about 4.7 kilograms fresh weight of fish. If you assume one generation of organisms per year, this would say that perhaps you are making ten additional pounds of fish available, or you are increasing the total fish biomass in the lake by 10 pounds for the years time. It's easy to shoot this full of holes, but it gives a potential, it's sort of a maximum. We are thinking only in terms of food for fish and not considering very important additional factors such as reproductive impacts, which we did not even attempt to assess, and I suspect they are probably more important than what we are talking about here. But to me, 10 pounds of fish was sort of a surprising find, in that it wasn't nearly as much as I expected it might be from a given outfall. If someone were talking to me about the enhancement value of an outfall as opposed to the toxic effects that we might have seen, this was one of the most extreme effects. It was not really as high as I had expected. Maybe I had my sights set a little bit too low.

I would like to leave off there and would welcome someone shooting this full of holes, simply because it would stimulate some discussion in this realm. We very desperately need to find a good approach to this kind of evaluation. If there is someone using, say multi-varied analysis, to try to coordinate their observations of biotic and abiotic factors at community distributions versus, say, presence or absence of taxicants of various types, I would like to know if there has been some success with this, because we're very heavily into this sort of thing. We are getting very strongly into the measurement of impact, the presence or absence of organic toxicants and we are going to be hopefully extending the types of studies that I'm talking about here to that realm. We desperately need this sort of information.

2) Mr. McQuivey

I'm going to briefly discuss some of the results that we have obtained from a sediment modeling on the Cuyahoga River which was one of the areas that we found a potential problem. Many of the problems were sediment related and, as such, we really wanted to define the fate and the effect of sediments using mathematical models. Our study was conducted in conjunction with Colorado State University and the objective was really to determine the feasibility of determining the fate of CSO materials using models. We selected the Cuyahoga River at Old Portage down to Lake Erie primarily because there was a DO problem that we had recognized in an earlier study, plus the U.S. Geological Survey had collected many years of sediment data at two locations. It was one of the areas that we felt had enough good historical data that we could make a fairly good attempt at using the model.

The principles of the sediment modeling was a two step process. We used an unsteady flow model that Dr. Keefer had developed while he was with the U.S. Geological Survey and we coupled that with a sediment transport model developed at Colorado State University. It was a watershed model and we modified and coupled that with the flow model which was one dimensional. The advantage of this type of model is that it allows you to look at what happens in space and time and there is a great deal of things that we could talk about in terms of the flow model and sediment model and the data needs.

Our reach is just below Akron, Ohio down to Lake Erie, about 40 miles. This gives you an idea of some of the things that we found in terms of the DO study as well as some of the observed stages and discharges as well as the modeled results there. The backwater profile at a given flow condition is a useful technique that we developed to calibrate our flow model and the sediment model. We're interested in the deposition and scour at different locations along the river. This model was set up to take particles in the sand, sizes .06 millimeters on up, and the material was either scoured or deposited at different places along the river in the longitudinal direction. In a stormwater runoff bypass below the Akron sewage treatment plant there are two little sediment traps in the reach of the river. Little impoundments trap much of the material as you route it downstream, and you can change the bed elevation, Delta Z. It is depositing and then when the flow comes through down there, within minutes the materal is scoured.

We also did some other things where we were concerned with heavy metals dealing with the fact that there is a specific gravity variation in materials that are deposited. What happens to heavy metals as they are input into receiving waters? Where do they deposit and how are they scoured? I think we used a five year storm event. Most of the sand sizes were scoured and deposited down in Lake Erie. The Corps of Engineers dredges there every year to keep the channel open. Most of the heavy metals settled out in these two sediment traps. Even with the five year event they did not go through the system.

As you probably know there is not a lot of sediment data collected. I think the U.S. Geological Survey has about 1,000 sediment stations. You can get daily data from about half of those. Some of them are weekly, monthly, semi-annual grab type samples. Sediment data is expensive to collect and therefore in terms of using mathematical models in fate and effect of sediments, it is probably not likely that the location is going to be exactly where you are going to have good data. There are only a few of those places around the country. Data is probably one of the most critical issues in modeling sediment transport. Perhaps we need to collect a good set of data below a CSO and verify the model a little better than we did in this particular case. There were some shortcomings. We have no data from the stormwater bypass below the Akron sewage treatment plant.

I have great faith in the model. We have been applying this for the Corps of Engineers and various other people, in terms of determining where to dredge. It is something you are going to have to do annually, and we've been, more or less applying it to situations like that, except in terms of CSO materials. I think that without data we are only going to get a qualitative feeling for what is happening. With good data I think we can put some quantitative numbers on the results. In terms of the research requirements we need to better verify the sediment models in terms of CSO materials. Even though we feel comfortable because they have been applied for years in different circumstances, we need to collect a good set of data and verify these models. Most sediment models have been developed primarily for the sand sizes, and in CSO's we have materials that aren't necessarily spherical in size. There is a broad range of materials that most of the present sediment models do not take into account, so one of the things we need to do is to extend these models to the fines, the cohesives as well as to other particulate shapes, fall velocities and what not.

3) Mr. Mancini

I would like to divide my remarks into two portions. The first part of my comments will address what I see as general research needs, and the second part will concentrate on more detail in the mathematical modeling area. The perspective that I make these comments from is as follows. I see that the urban runoff and CSO program, as proposed, will cost the taxpayers of this country from 40 to 80 billion dollars in capital cost. I would gather that over the lifetime of these projects that cost might be tripled in terms of both capital and operating costs. These expenditures are being looked at in an environment and a period where heady concern for water quality has abated. The environmental concerns are beginning to face the realities of competition in an open society with health, defense, education, energy and the economy, and it seems to me that our profession is facing a crisis where the survival of the remnants of the environment movement are really at stake in the long run. I think, in addition to these factors, there is an assidious force that may be driving our profession. Toxics has become the word of the decade. Let me relate to you the experience I had as a professional.

Ten years ago the word of the decade was thermal pollution, and in many places, thermal pollution was a very real environmental threat. Hundreds of millions of dollars were spent on studies of thermal pollution. Billions of dollars were spent on the abatement of thermal pollution. Thermal pollution has not been mentioned in an environmental conference that I have attended in five years because it has turned out to be a very site specific problem: a real concern at special locations, but a non-problem across the nation. I would submit to you that toxics may inherently have the same characteristics.

At the present time toxics are the concern of the decade, and they are clearly of vital concern with certain types of materials and in certain environmental settings. Clearly,a dump of keptone in the James River that contaminates the river, the fisheries and has health implications is a significant environmental concern with respect to toxics. The same thing is true with PCB's in the Hudson and Escambia Bay. But, do toxics really pervade all of our environmental concerns? I submit to you that the jury is still out, but there is a real possibility that toxics may be overblown as to importance in our kind of thinking and our research people ought to be very aware of this in terms of their guidance of our research efforts. Therefore, I suggest that our research efforts should maintain a balance. They should clearly continue to address the toxics problem which has a lot of money being poured in from other areas in addition to the municipal branch. The municipal branch ought to kick in their share of the pie, but I think it would be a grave mistake for the research and development efforts in the municipal area, and in particular in the CSO and runoff area, to concentrate too large a percentage of their efforts in the toxic area to the exclusion and ignorance of the conventional pollutants.

Getting back to the first point that I made that the climate in which we are operating as environmentalists is changing. There are two very significant examples that highlight this. In the first instance, EPA is beginning to require and request information in the CSO program, which is a funded construction grants program by Congress for a benefit analysis of CSO before they spend capital. So, the times are changing. We are no longer in the era where we are willing to afford expenditures for environmental control without regard to benefits. The agency, being a representative of the administrative branch of government, is, in fact, reflecting the public attitude and beginning to require a benefits analysis with CSO. Even a closer reflection of this public attitude is Congress' attitude as indicated by Dennis Athayde. They essentially said, "Hey, is urban runoff a national problem; and, will you define for me, Congress, the problem in terms of impairment of beneficial uses?" So, the handwriting is on the wall. We are now competing with all sorts of public programs for money. In addition, CSO and urban runoff are problems of national significance, or if they are problems of local significance, there are some very specific questions that we are being forced to answer now. The pressure to give more definitive answers to these questions will increase with time. As an example of the questions: Does this type of waste, CSO, urban runoff and so forth have to be removed? If so, at what locations? Every single pipe, every other pipe, one in ten pipes, a hundred pipes? How large a facility or action do I have to take to remove it? Should I remove 10% of the total annual load? Should I remove all of the small frequent events and take a beating once every year? Five times a year? Once every ten years? I think these are very significant questions because they change the order of the 20 to 60 to 80 billion dollars. If you design for the frequent small events you will change your order of expenditures.

Finally, what type of trade-off exists? I've got to say I'll give 10 pounds of fish for 4 million dollars. You know those are the kind of decisions, if this profession is going to survive, that I think we are going to have to make. Our research efforts have to be directed towards providing those answers for both conventional pollutants as well as the toxicants.

Now, let me be a little more specific in terms of the general research needs. It seems to me the impairment of beneficial uses is a real area of opportunity in terms of a research program. In particular, it seems there is a need to define what is the beneficial use, and to define a beneficial use at several levels. So it's not just enough to say fisheries are a beneficial use. We have got to define that at a number of levels so that we can begin to consider tradeoffs. We can do that in terms of dollars, in terms of fish population, or we can do it implicitely in terms of standards, but we have to come to grips with that problem if we are going to be viable as a profession. Finally, we have to develop methods of assessing benefits where there are competing resources. As an illustration - many urban runoff problems and CSO problems are confined to very small streams, this is particularly true with the urban runoff, very small streams within cities. Surrounding these cities may be very luxurious alternate resources. Beautiful lakes with large fisheries and so forth. What is it worth to clean up the little stream when the population has an alternate resource which is much more desirable? These are very tough judgements and we as technologists have to have input from the information standpoint to the public and the elected officials, otherwise they are going to make the decisions without the knowledge and we are going to cry, one way or another. If we solve all of these problems in terms of beneficial uses, we have to translate those into methodologies that can be applied without spending 100 million dollars on each site to determine what we should do on that one site. So, the methodologies have to be condensed and simplified so that they can be applied on a site specific basis and answer some of these real problems with respect to effects.

It seems to me that there is a need to look at effects for combined overflows and urban runoff, acute and chronic effects. We might want to consider looking at these biological effects in two ways. The first is by the individual components, where we can go back to the literature and pick off much of the historical data base; that is, cadmium, LC 50's, measurements of this, that, and the other organism, and for fish and the lake. On the other hand, it might be worthwhile to sponsor an effort which would look at "urban runoff and CSO's in their totality" to see what the effects of the total combination would be. Some remarks have been made here about cynogistic effects, antagonistic effects, and they may need to be second order, but are not there. It seems to me that some screening research money to look at effects, both chronic and acute, from urban runoff and CSO's are needed in this area because we have a very unique problem in the stormwater area. We had 88 days of rain in Austin, Texas. That's almost a desert for pete's sake. So there is a lot of little shots of either toxic effects or low DO's and so on. The chronic impacts may be significant. On the other hand, there are some very large events in which the acute effects may be very significant. We as a society, have to consider devoting some resources to looking at both of those aspects.

If we do determine some areas where there are effects, we have to be able to synthesize this fact in terms of estimating populations. When we think of events that are essentially a telegraph wave of various side events that occur in time; we get a rain and it's dry, we get a rain, and it's dry. Then factors like recruitment and regrowth of the population are very significant so that if we get a 10% shot each time it rains in terms of mortality or a destruction of the population, but in the three to five days between rain storms the population can revitalize itself back to equal zero or 100%, then we really don't have a water quality problem. So, there are whole areas that have to be addressed in terms of coalescing the laboratory work with theories that will allow us to understand recruitment issues, regrowth issues and population issues, areas which will dovetail with our ability to look at benefits of biological effects.

Another significant point is the fate of the materials from the CSO's. Let me point out something that Jim Heaney called to my attention several years ago and I really grabbed onto this as a nice analogy. If you stop and think about it, there are two types of effects from CSO's and urban runoff discharges. There are the primary, or immediate effects so that you take an urban runoff or a CSO discharge and there is some little critter in there, and he gets hit with a low DO or copper cadmium concentration that's high and he responds - he dies, or lives, or gets sick - something like that, but it is an immediate effect. The storm has passed, the critter begins to revive and return to the original state depending on what the effect was. That is the primary effect of the CSO discharge. The primary effects are applicable for conventional pollutants as well as for toxic materials, or nonconventional pollutants.

Then there is a whole series of secondary effects which are associated with the bottom deposits and things of that nature. You hit a body of water with a CSO or urban runoff and you get deposition of material. Again, it's appropriate for both the conventional and non-conventional pollutants. I'll give you two specific examples: dissolved oxygen bottom demand, very important in many situations, often created by CSO's and urban runoff, and often created by algae settling upstream or runoff from agricultural areas. Many causes, but that is a long term assidious effect that could be associated with the two factors that we are dealing with. Also, the fate of toxics, particularly cadmium, is unknown. It may become part of a lignent system absorbed on the particulate or is chelated by organics and eventually drops to the bottom. What happens to it in time? Anaerobic conditions may occur, the redox potential may change, and so we may get a resoluabilization of the heavy metal or the organic material and there could be a long term effect of the same load. So, there are two aspects of the CSO and urban runoff, the primary effect, and then this long term subtle impact. It seems that we should consider committing vital research money towards defining the fate of these contaminants, both toxic and conventional pollutants. When we start putting in all of these wonderful devices, we've got to monitor compliance with them. So, some effort is necessary to begin to decide how we're going to monitor and whether we're making it or not. Do we have to increase the amount of capital and operating costs we are putting in? Do we have an opportunity to reduce it? We need monitoring on a wholly different time and space scale than we do for the continuous discharges.

Given this list of effects and benefits, how successful are we in our treatment devices in terms of really handling all of these toxics and conventional pollutants? Can we make it in terms of this benefit analysis with the existing technology for the treatment devices? All of these research efforts require either lab studies, field studies, or model calibrations. This profession is in a changing climate for environmental control and we are being required to justify and demonstrate the benefits from expenditures that we recommend. We need to organize our research program to provide packets of information that fit into an overall mosaic that allows us to answer some of these questions. Looking backwards, CSO and urban runoff programs have been very successful. They are one of the shining examples of EPA research which led our profession in the treatment and load assessment area from zero, knowing nothing about effectiveness of treatment and a real estimate of loads from CSO's and urban runoff, to the point where we can consider a whole array of treatment devices and make some pretty subtle comments on the kinds of load variations that are occurring. Now this program is facing a new challenge to meet todays needs, different needs. They are in terms of benefits, effects, and fate, and if these guys can respond to that they will have done the agency, the public, and our profession a service.

4) Dr. Wanielista

I would like to amplify on John's comments and the underlying theme of the conference which essentially is receiving water quality responses and expand on it to indicate receiving critter responses and biological responses in addition to the water quality parameters that we measure. Yesterday and the day before we heard some talks on lakes and lake responses. I want to specifically limit my comments to a project which is ongoing in this local area. One that Harvey Harper has responded and talked on, Lake Eola. Essentially, we are dealing with the storm sewer system and land-locked lake. The lake itself is about 27 acres and the watershed is only about 131 acres. We are in a fortunate position because it appeared that the problem was very well defined to start out with; that is, from the Mayor's office right down through public works, citizen groups and everything. There was a problem with the odor out of the lake and the aesthetic condition of the lake, not to mention anything about water quality. There was certainly a political move to do something in order to keep the politicians in office, and they were very responsive to the needs of the people. Joggers complained. There were numerous cultural events, bandshell concerts and craft events. Over 400 craft booths spread around the lake in which people actually paid to put a craft booth there and sell their wares during the show. There are two of these, called Fiesta in the Park, which go on at certain parts of the year and then are extended to other times of the year depending on the demand. So, there was indeed an economic benefit as the city saw it. There were also children's benefit programs, fishathons and what not, that were being conducted and essentially the city was very interested in protecting this and doing something further.

In 1972 we did a brief benefit cost analysis which had nothing to do with the craft shows and everything else which is going on, but had to do with the land values around the lake and what would happen to the land values. Or, how can you compare this urban setting to another urban setting where you had a difference in the water quality for similar activities. I think you'll recognize the value of land in the downtown area and essentially you could add up the dollars and find out that it would be significant. It was really an unscientific study which was based on calls to realtors to find out what their opinion was. We came up with a figure which was over one million dollars for the land value, if indeed, the lake went to an odorous condition.

There are a lot of other ways of doing benefit cost studies. My own experience in recreational values say that it's very difficult, and you have to come up with some very well designed programs. For example, on the Lake Eola thing, there are somewhere in the neighborhood of 2,000 people that go there over 30 times a year to visit a concert. There is a concert which is held for two hours. It is supported by the local newspaper and the city of Orlando. There is a dollar value attached to this. People actually expend energy, time, money and everything else to get down there and listen to school bands, Navy band, University band, art performances and what not. Some things are quantifiable and I'm saying this because I think there was justification for going into and looking at a restoration of this particular lake.

The program we designed had a lot of detail in it and we used models. One of the earliest models we used was a time series model. We tried to relate the responses of phosphate in the lake to discharges of stormwater. We actually came up with some numbers and some relationships and I would say that it was called guessing numbers and fitting the data at that time. But, at least it gave us some kind of indication that we might have a problem here and maybe we could amplify and get some more data. Our own studies then went into the bioassays which were presented to you. We also did the eutrophication modeling, the Vollenweider model, Dillion models, Larson, Shannon Brezonik Index and all of these kinds of things which indicated that the lake was eutrophic to a certain degree. This gave us an indication that possibly we should remove some phosphorus from the lake. The bioassay studies gave us an additional indication that indeed, the biomass responses seemed to be stimulated by the additions of stormwater. We noted some toxic effect and the visual observation of the situation gave additional information after a storm event. If you went out and just visually looked at the system, you would probably find out that there was some response - that the algal blooms tended to be associated with the rainy season and they tended to be more frequent during that time period.

One thing we did that was interesting was to go in and examine the runoff. Over 50% of the nitrogen and phosphorus was dissolved in the actual runoff collected going into the lake. This would limit your technology for best management practice. If you're talking about technology, which essentially is only dealing with sedimentation or physical removal, you may limit your choice of the particular management practice; also, the particular compound or form that you're looking at. Does this particular form actually produce a response in the algae blooms that we were getting on the lake? So comparing the existing models, the studies that we did, and everything else led us to a management scheme where we are diverting the first quarter to a half inch of runoff. Probably from about 0.9 to 1 inch of rainfall in that particular area. This is based on a mass removal that we have done over a long period of time. We actually went out and measured these diversion systems, found out what the diversion volumes were, the quality in the diversion volumes, and then went to a stochastic simulation using stochastic rainfall traces over a 20 year period and tried to estimate what the mass diversion would be and then come up with frequency distributions on the mass discharging into receiving bodies of water. This all resulted in an implementation plan which we're currently undergoing.

This project wasn't too hard because we had a lot of help all the way from U.S. EPA Storm and Combined Sewer Program, 314 Clean Lake Program, Region 4, Legislature, Tallahassee DER, down into the local level and everybody got together and recognized the problem. The funding level for the actual implementation actually came from the State Legislature who was friendly toward the project. They came up with a big chunk of money, $190,000 from the legislative pot. The city and the university also provided a small matching amount and took that money and went to a 314 Program. To try to get in you must get everything together and try to do something about the runoff into the lake system. You should look at the receiving water body and possibly make some adjustments within the receiving water body itself. Lake Eola didn't have very much rooted vegetation. It didn't look like there was an "equal system" set up to assimilate and recycle the material that will have to be changed in some ways to at least provide more of a natural system for similation. This is also acceptable to the people of the area. They are the driving force. They would like to see all of the natural area returned to the downtown area of the city. I believe benefit studies can be done. They're expensive and they can be done in a lot of detail or in a little less detail. I believe that the driving force on many of our projects is not so much what the water quality is but what the people want, and that has been brought up quite a few times. As engineers and scientists we should be responsive to that need.

5) Mr. Moffa

I want to touch on one element that was brought up twice and that is public participation. We're talking benefit. I don't think there is any question that we need the numbers to define the benefit, but most important appears to be justifying what we're going to be doing since we surpassed the secondary treatment stage. That was really mandated by the 1972 Amendments. We have finally come to the point where we are riding out that crest and facing a new challenge. We environmentalists had better show that we are really doing some good before we go any further because we have some very real competitors. Energy is probably the one that is going to be very much the lead in some of our priority issues. We've got to talk to some of the people in the community that are going to be receiving the benefit of the expenditures we're proposing. This is a whole new ball game and frankly we engineers are totally inept in this area. We've got to bring in the communications experts. We've got to make it simple enough so it's understood. We can no longer say, "this is mandated, we've got to do it." Even when we could say that, it was a challenge. We are reaching the point where the regulatory agencies are saying that we need justification and I am glad to see it having been involved in a number of projects where we studied the impact after the facilities were designed. It really isn't the way to go. So, I think, we've got to begin to bring together the forces needed to heighten our public participation and set our objectives before we go any further. In a way it might be a plug for 208, although that seems to be running out of funds.

Mr. Bob Foxen, U.S. EPA, Arlington, VA

I agree with a lot of the things that John said about the need to consider a cost benefit in deciding what kind of treatment we want to provide. A problem is the way the law is structured now. It precludes looking at weighing cost versus benefits in deciding what kind of treatment is required, whether it's AWT treatment or CSO. The law requires that we achieve fishable, swimmable waters where attainable and as I see it, there is not room legally for us to decide whether it's worth 10,000,000 dollars, whether 10 pounds of fish are worth $10,000,000, so it seems that the answer is that there is some kind of legal change that is required before we can adequately look and weigh the costs and benefits in deciding what level of treatment is needed. Karen, perhaps you could describe your program.

Ms. Karen Klima, U.S. EPA, Washington, D.C.

Bob and I worked on the same program. I'm working on the AWT reviews and the issue there is similar to CSO control because it's a situation where secondary treatment may not be adequate to meet the water quality goals or to achieve the beneficial uses. But, in the AWT review, we're not deciding whether the cost of advanced treatment is worth the investment. We are not deciding whetherit is worth the 10,000,000 dollars to add on additional filters and whether the benefit in terms of a fishing use is worth that investment. We are assumming that the fishable, swimmable use, is a goal we're shooting for. I don't necessarily agree that it is the way to go, but I know for a fact that we've had arguments inside the agency to the effect that we cannot weigh cost versus benefits in deciding whether this 10 million dollar investment is justified.

I have no other interpretation, I can just say that we are talking about the subtle distinction between cost benefit analysis as opposed to cost effectiveness analysis that we have been doing in our AWT review. Bob and I feel like we don't have the legislative reform to really be looking at cost versus benefit, but we can look at cost effectiveness now. As to how that distinction applies to the CSO benefits that we are looking at, let me suggest this as one possible answer. If that is the case and the agency does decide it is a mandate from Congress, you are very fortunate because that gives your research program time to anticipate that the next level of question is going to be, "What am I getting for my dollar," rather than, "How effectively am I spending my dollar?" Maybe you can use that time effectively to get those answers. I don't know whether the agency position is right or wrong. I think it's a subtle distinction that really isn't in the public's best interest.

Dr. Fred Lee, Colorado State University, Fort Collins, CO

There is a real undercurrent to shoot down inappropriate standards in a very practical way. It is not just us academic and technical guys that are talking to each other. I know some guys that are even thinking about putting up some money to alter standards.

Mr. Phil Graham, U.S. EPA, Washington, D.C.

PG61 was a first effort to look at the benefits because we thought CSO's were going to be so expensive. Most of the AWT reviews are not CSO projects, although a few of them were. Some of them involved sewer separation where it had gone so far that it was too late to even worry about it. I think if we clarify it for the treatment plants, that's probably the agency policy and PG61 is also the agency policy on CSO's. Its handbook coming out in the future will probably say that the way to bring urban stormwater into it, you have to consider the effects of it in your planning for both point sources and CSO to make up for what you can't do. That's not eligible in urban runoff so I don't know what will happen with that.

Dr. Mike Sonnen,

The people from EPA are not quite correct. They do have the legislative mandate to do benefit analysis. It was in the 1977 act and some before that but it hasn't changed lately. The numbers are Section 302B2, if you would like to look it up, and it has to do with the administrator being warned in advance by Congress that, if certain dischargers can show that the benefits of putting in advanced treatment are not as great as the cost that the administrator will have to take, then take a second look at the discharge requirements for that discharger. This is an area that I have worked in since 1967, so I know that benefit analysis has been done and EPA has had the legislative wherewithall to do it. They have also run into problems with the economy lately and I have had people from EPA say, "Yes, but we're just not going to enforce that section." Well, that's a different problem. To say that they don't have the mandate is not true.

Mr. Dennis Athayde, U.S. EPA, Washington, DC

I would like to stress two points that John Mancini made. One of them was kind of hidden and the other one was obvious. The one that was obvious to me was the need for us to define the relationship between combined sewer overflow and separate storm sewer discharges. If we are going to enter into a controlled program for combined sewer overflows, we can't ignore the separate storm sewered areas in these cities. We are going to have to have some optimum type program. That is one of our needs.

The other thing I would like to quickly touch on is the use of models. How are they ultimately used? How was their output used? The decision-makers use models as a tool to try and help them make better decisions. Ultimately, the decisions are not made from the output of the computer, they are made politically. We understand that and we are not going to worry about the level of detail or the complexity of our models. We are going to go for the simplest one that will give us an answer and allow us to make a good decision.

Mr. Ken Weldon, Florida Department of Transportation, Tallahassee, FL

By virtue of that, I have a slightly different outlook than some of you. I came to hear some definitives regarding various methodologies for handling some of the problems due to the impact of receiving waters. We are faced with a situation of being able to obtain permits through DER, EPA, and so forth. So what happens in our case? We run into a problem, not only with terminology, but in a general overall sense of a lack of understanding in certain areas of hydraulics and hydrology. For instance, I think we need a definitive set of standards for non-CSO's as has already been mentioned. Secondly, we need to assess the actual condition or standards for the location. There are many areas in which we are invloved where we are accused of creating an impact in which the existing area is undeveloped and by virtue of its characteristics is worse off at times than some of the urban areas. We need to establish a norm and I think it should be done on a point basis because too often we're spending too much of the taxpayers money in areas in which we don't need to. Again, we get back to cost benefits.

In using these models for determination, such as was this a stream, lake, or what-have-you, I think we need to give more study to the idea of the hydrology of the situation. For instance, I know several times the frequency analysis was mentioned and I'm quite amazed at some of the input that is used. I mentioned that to Dr. Lee, and I realize that often times he doesn't consider the frequencies, but one of the things mentioned this morning was a five year storm used to determine a certain impact. You may start out with a five year storm but you don't know if you are going to end up with a 25 year event at the particular site or location. We need to be able to use more control points in these modeling situations if we are going to ascertain anything at all. There has to be sufficient data before it is ever attempted to establish what the norm is for a specific location and I think the norm of the location should be used primarily for the calibration of these models.

In regards to research, development should be made for establishing methodologies for input. I'm involved quite often in trying to design outfalls, or storm sewer systems which naturally enter into lakes. There is no way you are going to prevent it. They are existing situations and we're going to have them until the end of time. We're going to have to develop some kind of terminology, some design, whereby certain things are described, again methodology due to velocities. Sediment transport seems to be one of the major problems. They are stirring up the bottom deposit and again we get right back to what is the norm. I want to encourage someone to come up with some definitive classification for types and bodies of waters. For types, I don't care whether you want to consider a developed area around a swamp or streams just remember - they are not the same and shouldn't be considered the same. It seems like everything we have is on an idealistic basis. I just don't thing it applies.

Dr. Dennis Lai, Clinton Bogert Assoc., E. Brunswick, NJ

I want to explain Dennis' comment that we should not separate combined sewer overflow abatement from storm sewer discharge. The annual average discharge from the combined sewer area is very much comparable with the separate sewer area. We feel that it doesn't make sense to abate combined sewer overflow without abatement of stormwater discharge from the separate sewer area, so we are proceeding with the project and hope to make a case to EPA to fund separate sewer area abatement. I hope that in other parts of the country where they have separate sewer systems, they find the same situations and we can communicate with them. The point is to encourage seeking out the relationship on site specific basis between CSO loads and their impacts in urban runoff.

URBAN STORMWATER AND COMBINED SEWER OVERFLOW-IMPACT ON RECEIVING WATER BODIES

EPA NATIONAL CONFERENCE
ORLANDO, FLORIDA
NOV. 26-28, 1979

Summary of

Workshop on Practical Applications and Research Needs for Receiving Water Responses to Urban Stormwater

Session I

by

James P. Heaney, Ph.D., P.E.
Dept. of Environmental Engineering Sciences
University of Florida
Gainesville, Florida 32611

The opening session was divided into two major activities: opening statements by panelists, and discussion from the floor. A summary of the comments, not necessarily in order of importance, is presented below:

1) Documented evidence of receiving water impacts from urban stormwater is scarce.

2) Existing stream standards are inappropriate for incorporating stormwater impacts. An overall response in terms of a frequency distribution would be an improvement.

3) Receiving water quality "problems" need to be related to impaired beneficial uses and whether installing controls will enhance benefits. While a case-by-case approach is preferred to uniform standards, it remains to be seen whether such an approach is practical from an administrative viewpoint.

4) More formal procedures for calibrating and verifying simulation models are needed. Included in such procedures would be guidelines for better matching the model to the data base.

5) The importance of benthal deposits in analyzing wet-weather problems was stressed repeatedly.

6) More effort is needed in using simple biological indicators to examine receiving waters for planning purposes.

7) Fundamental research is needed on deoxygenation and reaeration kinetics.

8) Information on health effects of stormwater needs to be gathered even though it is expensive.

Overall, there seemed to be a consensus that we are returning to the earlier (pre-EPA) philosophy of environmental quality management wherein the orientation was towards public health and the burden of proof is partially on the regulator to justify that the proposed control program is cost-effective.

Session II

by

John L. Mancini
Manhattan College
Bronx, New York

Receiving Water Impact Research Needs:

1) There is a need for research and development programs to maintain a balance between projects which address conventional and/or classical pollutant variables and those projects that consider some of the issues associated with toxics and heavy metals. This requirement is particularly significant in the area of impacts from intermittent urban discharges such as CSO and urban runoff. Programs for construction grants for facilities are entering the heavy expenditure stages for CSO and are under evaluation in the urban runoff area. Analysis of the water quality impacts of these discharges and identifications of the effects on water usage are currently being required and will become more critical in the future as the large size of expenditures and tenuous nature of impacts becomes more evident.

2) A specific research and development need is found associated with relating the size of urban areas and receiving water bodies to distinguish situations in which there is a reasonable probability of encountering a water quality problem from those where the existence of a water quality problem is a more remote possibility. The definition of what constitutes a water quality problem is beginning to focus on water use interference and therefore research and development projects can be specifically directed towards this definition of water quality problems. The assessment of water quality problems, using interference with water usage as an operative definition, will also provide a framework in which estimates of benefits associated with treatment of CSO and urban runoff can begin to be developed.

3) EPA research and development programs should, in the opinion of the writer, begin to address the areas of water quality problem identification and benefits assessment. Expenditures for environmental control have begun to

encounter increasing competition on the federal, state, and local levels from other programs. In order to retain the momentum of the national drive for environmental quality it will be necessary to identify tangible benefits to society. Research efforts in this respect should consider projects in each of the following areas:

a) Studies of the fate and transformation of pollutants in natural environmental systems. Analysis should be related to the types of receiving water systems, such as streams, lakes, estuaries and oceans and must also consider the unique characteristics of urban discharges.

b) Particular attention needs to be directed to the solids associated with urban discharges. The transport and interactions of particulates and associated pollutants are key elements in evaluating impacts or urban discharges.

c) Investigation of effects of contaminants associated with urban runoff represents fruitful areas of research activity which can be carried out conjunctively with other EPA programs in toxics, monitoring and standards. Particular emphasis should be directed towards using information from these other programs to build a base of knowledge which can be employed to address the unique aspects of urban discharges associated with the frequent yet intermittent nature of these impacts.

URBAN STORMWATER AND COMBINED SEWER OVERFLOW
Impact on Receiving Water Bodies

LIST OF PARTICIPANTS

Alvarado, Luis C.
MDC
59 Selden Hill Dr.
West Hartford, CT 06107

Ammon, Douglas C.
U.S. EPA
Edison, NJ 08817

Anderson, Craig P.
Jefferson Parish Env.
3600 Jefferson Highway
Jefferson, LA 70121

App, Charles W.
U.S. EPA
716 Raynham Rd.
Collegeville, PA 19426

Athayde, Dennis
U.S. EPA
Washington, DC 20460

Badalamenti, Salvatore
U.S. EPA
26 Federal Plaza
New York, NY 10007

Bain, Richard C.
Brown & Caldwell
100 West Harrison
Seattle, WA 98119

Barfield, Larry D.
Fla. Dept. of Transportation
2630-A Old Bainbridge Rd.
Tallahassee, FL 32303

Belanger, Thomas V.
Florida Institute of Technology
112 W. Peekskill Pl
Melbourne, FL 32907

Bell, John M.
Purdue University
3 Concord Place
Lafayette, IN 47905

Bennett, Francis P.
Retired
14103 Rutland
Detroit, MI 48227

Berger, Arthur W.
City of Painesville
7616 Mountain Park Dr.
Mentor, OH 44060

Birkitt, Beverly F.
Dept. of Environmental Regulation
1907 Sherwood Dr.
Tallahassee, FL 32303

Blancher, Eldon C.
University of Florida
Gainesville, FL 32601

Boyd, Gail B.
Woodward-Clyde Consultants
3 Embarcadero Center Suite 700
San Francisco, CA 94111

Bozeman, Martin
Woodward-Clyde Consultants
3 Embarcadero Center Suite 700
San Francisco, CA 94111

Branch, James C.
Clark, Dietz Engineers, Inc.
500 W. Fulton St.
Sanford, FL 32711

Brill, Edward L.
Floyd Browne Associ. Ltd.
750 Sable Court
Youngstown, OH 44512

Brown, E. Ryland
and Community Development
N.C. Dept. of Natural Resources
Raleigh, NC 27611

Brownley, Dennis D.
U.S. Air Force
USAF Clinic
Randolph AFB, TX 78148

Brunner, Carl
U.S. EPA
Washington, DC 20460

Burnett, William C.
Florida State University
Dept. of Oceanography
Tallahassee, FL 32306

Byrne, Christian J.
FSU-Oceanography
1900 Nininger St. #7
Tallahassee, FL 32304

Calabrese, Mark
Reynolds, Smith, & Hills, Inc.
7120 Lake Ellenor Dr.
Orlando, FL 32856

Canale, Raymond P.
University of Michigan
Ann Arbor, MI 48109

Cavicchi, Paul
NH WS & PCC
3 Penwood
Penacook, NH 03303

Celikkol, Barbaros
University of New Hampshire
P.O. Box 361
Durham, NH 03824

Chojnowski, Kathy

Church, Paula H.
Dade County Dept. Env. Res. Mgmt.
909 SE 1st Ave
Miami, FL 33131

Clarke, Bill
Proctor & Redfern, Toronto
9 Cedar Ave.
Toronto, Ontario, M4E1K1

Clarkson, Robert A.
Calocerinos & Spina
1020 Seventh North St.
Liverpool, NY 13088

Cole, Michael S.
City of Syracuse Water Division
141 Hickok Ave.
Syracuse, NY 13206

Condon, Francis J.
U.S. EPA
Washington, DC 20460

Courtney, Charles M.
The Delton Corp.
990 N. Barfield Dr.
Marco Island, FL 33937

DeZolt, Jim

Diniz, Elvidio V.
Espey Huston & Assoc.
2500 Louisiana NE #310
Albuquerque, NM 87110

Diorio, Gary J.
Floyd Browne Assoc. Ltd.
4631 New England Blvd.
Youngstown, OH 44512

DiToro, Dominic M.
Environmental Engr. & Sci. Prog.
Manhattan College
Bronx, NY 10471

Donahue, Charles R.
Dept. of Oceanography-FSU
1305 Pullen Rd.
Tallahassee, FL 32306

Duba, George A.
Western Michigan University
220 S. Kendall #9
Kalamazoo, MI 49007

Duvall, Leland R.
U.S. EPA
6021 W. 99th
Overland Park, KS

Ellis, Franklin W.
CH2M Hill
Reston, VA 22090

Elson, Wayne D.
U.S. EPA
1514 Hill Ave. #2
Wheaton, IL 60187

Evink, Gary L.
Fla. Dept. of Transportation
3419 Thresher Dr.
Tallahassee, FL 32303

Falco, James W.
U.S. EPA
225 Hampton Ct.
Athens, GA 30605

Fancher, Dick
Dept. of Environmental Regulation
6336 Harvard Ct.
Pensacola, FL 32504

Field, Richard
U.S. EPA
Edison, NJ 08817

Fisher, Anthony P.
U.S. EPA
80 Hayward St.
Yonkers, NY 10701

Flood, John J.
E.T. Killam Assoc., Inc.
R.D. #1 Roxiticus Rd.
Far Hills, NJ 07076

Foerster, Klaus E.
Metro Waste Control Comm.
8274 Rhode Island
Minneapolis, MN 55438

Foran, Frank
George F. Young, Inc.
6797 21st Wag. S.
St. Petersburg, FL 33712

Forger, Dan

Foxen, Robert J.
U.S. EPA
4300 Old Dominion Rd.
Arlington, VA 22207

Frahn, Kurt S.
Post, Buckley, Schuh & Jernigan
1351 Yulee Dr.
Clearwater, FL 33515

Franklin, Marvin A.
U.S. Geological Survey
3044 Godfrey Pl.
Tallahassee, FL 32308

Garie, Henry L.
Rutgers University
17 Myrtle Ave.
Dover, NJ 07801

Garlick, Dan
Dept. of Env. Regulation
515 Peck Ave.
Ft. Myers, FL 33907

Garrity, Rick
City of Tampa-DPW
Tampa Municipal Bldg.
Tampa, FL 33602

Gerard, David
Illinois EPA
2200 Churchill Rd.
Springfield, IL 62706

Gietz, R.J.
Regional Municipality of Ottawa
655 Shefford Rd.
Ottawa, Ontario, KIJ8G8

Glandon, Robert P.
Michigan State University
6227 Balfour
Lansing, MI 48910

Godlewski, Victor J.
Stottler Stagg and Ags.
7878 Shoals Dr., Apt. B
Orlando, FL 32807

Golding, Bernard L.
Howard, Needles et al.
8982 Islesworth Ct.
Orlando, FL 32811

Gouin, Denyse
Govn. Quebec
C.P. 852 St. Redempteur
P. Quebec, GOS3PW

Graham, Donald S.
University of Florida
Department of Civil Engineering
Gainesville, FL 32611

Green, Raymond F.
Martin Marietta Corp.
229 Lk. Seminary Cir.
Maitland, FL 32751

Green, Robert I.
Agri-Leis Corp.
P.O. Box 6232
Lakeland, FL 32803

Greenstein, Bary P.
Collier Cty. Bd. Cty. Comm.
1270 Cooper Dr.
Naples, FL 33940

Griffin, Thomas L.
U.S. EPA
9003 Forest Glen
Palos Park, IL 60464

Grimshaw, Herbert J.
Oklahoma Water Resource Board
2833 SW 86th St.
Oklahoma City, OK 73159

Grizzard, Thomas J.
Virginia Tech
Box 733
Manassas, VA 22110

Gruber, David A.

Halbert, Bruce E.
James F. MacLaren Ltd.
151 L'Amoreaux Dr. TH111
Baincourt Ontario, Canada

Hampson, Paul S.
U.S. Geological Survey
2600 Art Museum Dr.
Jacksonville, FL 32229

Harper, Harvey H.
University of Central Florida
CEES Department
Orlando, FL 32816

Hawkins, Robert A.
TenEch Environmental Const.
515 Park Ave.
Louisville, KY 40208

Heaney, James P.
University of Florida
Gainesville, FL 32601

Hentschel, Greg E.
Housel & Assoc., Inc.
202 W Bearss Ave., Suite 250
Tampa, FL 33612

Hermanson, George H.
CH2M Hill
P.O. Box 3317
Naples, FL 33939

Hever, Robert F.
Floyd Browne Assoc. Ltd.
5855 Lawnview St. NW
Canton, OH 44718

Hewitt, James L.
Espeg, Huston & Assoc.
2500 Louisiana NE #310
Albuquerque, NM 87110

Hoffman, Eva J.
University of Rhode Island
Oceanography-URI
Kingston, RI 02881

Holtkamp, Michael L.
SW Fla. Water Mgmt. Dist.
13404 Monte Carlo Ct. #48
Tampa, FL 33612

Hoyt, Dennis K.
Lamp, Rynearson & Assoc.
9290 W. Dodge Rd.
Omaha, NE 68114

Huber, Wayne C.
University of Florida
Gainesville, FL 32601

Hvitved-Jacobsen, Thorkild
Institute of Civil Engineering
Aalborg, DENMARK

James, William
McMaster University
127 Dalewood
Hamilton, Ontario, L8J4B8

Jenness, Frederick M.
Minnesota Pollution Control
1935 West Co. Rd. B2
Roseville, MN 55113

Joiner, Thomas E.
Joiner Engineering, Inc.
113 W. Savidge St.
Spring Lake, MI 49456

Jones, Ann R.
Colorado State University
Fort Collins, CO 80521

Jouseau, Marcel RG
Metropolitan Council
300 Metro Square
St. Paul, MN 55104

Jowett, James R.
U.S. EPA
5622 Eastbourne Dr.
Springfield, VA 22151

Kaufman, Herbert L.
Clinton Bogert Assoc.
188 W. Ramapo Ave.
Mahwah, NJ 07430

Keefer, Thomas N.
Sutron, Inc.
1709 Prelude Dr.
Vienna, VA 22180

Keller, David B.
Keller & Kirkpatrick, P.A.
10 Ridgedale Ave.
Florham Park, NJ 07932

Kersten, Robert D.
College of Engineering
University of Central Florida
Orlando, FL 32816

Klima, Karen S.
HQRS EPA
401 M St., SW
Washington, DC

Klingelhoefer, Ginger D.
Anne Arundel County
906 Primrose Rd.
Annapolis, MD 21403

Klos, Kenneth C.
Dept. of Environmental Regulation
3319 Maguire Blvd.
Orlando, FL 32803

Knauer, Gregory W.
Booker Assoc., Inc.
1822 Marriott Lane
Barnhart, MO 63012

Knoff, Robert M.
Burlington Cty. Planning Board
716 High St.
Burlington, NJ 08016

Koval, Peter R.
O'Brien & Gere Engineers Inc.
648 Beacon St.
Boston, MA 02215

Kreutzberger, Willian A.
Rexmord Environmental Research
2504 So. 14th
Milwaukee, WI 53215

Kutash, Bill
Dept. of Env. Regulation
3915 McKay Ave.
Tampa, FL 33609

Kynast, Frank
Am. Soc. of Civil Engr.
Walt Disney World
Orlando, FL

Lai, Dennis F.
Clinton Bogert Assoc.
18 Manton Ave.
E. Brunswick, NJ 08816

Landon, John C.
Post Buckley Schuh & Jernigan
400 E. Merritt Ave.
Merritt Island, FL 32952

Lartigue, Date
Proctor & Gamble Co.
Ivorydale Tech. Center
Cincinnati, OH 45217

Latch, Mark M.
Dept. of Environmental Regulation
1907 Sherwood Dr.
Tallahassee, FL 32303

LaZenby, Mack
City of Sanford
P.O. Box 1778
Sanford, FL 32771

Le, Tung T.
City of Tampa
3314 W. Louisiana Ave.
Tampa, FL 33614

Lee, G. Fred
Colorado State University
Fort Collins, CO 80521

Librach, Austan S.
Washington Council of Govts.
1875 Eye St., NW
Washington, DC 20006

Loijens, Harry S.
222 Queen St., 10th Floor
Ottawa, Ontario KIF5V9

Lombardo, Robert J.
Civil Engineering Consultant
825 4th St. West
Palmetto, FL 33561

Lopez, Miguel A.
U.S. Geological Survey
912 Tangelo Place
Brandon, FL 33511

Malueg, Kenneth W.
CERL-U.S. EPA
200 SW 35th St.
Corvallis, OR 97330

Mancini, John L.
Manhattan College
New York, NY

Mangan, Bob

Manz, Pete E.
Henningson, Durham & Richardson
8404 Indian Hills Dr.
Omaha, NE 68114

Markunas, Raymond
The Metropolitan District
51 Virginia Lane
Tolland, CT 06084

Markus, Ed

Marshall, Frank E.
Russell & Axon, Inc.
1803 Peninusla Ave.
New Smyrna Beach, FL 32069

McGinn, Joseph
MA Dept. of Env. Quality Engr.
70 W. Boylston St.
Worcester, MA 01606

McGrew, Thomas W.
Pinellas County 7607 Palmbrook Dr.
Tampa, FL 33615

McLellon, Waldron M.
CEES Department
University of Central Florida
Orlando, FL 32816

Medina, Miguel A.
Duke University
Durham, NC 27706

Meinholz, Thomas L.
EcolSciences, Inc.
Milwaukee, WI

Merrick, S. Preston
Reedy Creek Imp. Dist.
6 W. Preston St.
Orlando, FL 32804

Messenger, Allen
Texas A & M University
College Station, TX 77840

Miertschin, James D.
Espey, Huston & Assoc., Inc.
3010 South Lamar
Austin, TX 78704

Mikalsen, Karsten T.
Georgia Env. Protection Div.
3326 Wheeler Dr.
Atlanta, GA 30340

Milgram, Otto
E.T. Kellam Associates, Inc.
2375 Monica Pl.
Scotch Plains, NJ 07076

Miller, Lynn
SW Fla. Water Mgmt. Dist.
5060 U.S. Hwy. 41 South
Brooksville, FL 33512

Miller, Robert A.
U.S. Geological Survey
109 Bagberry Rd.
Longwood, FL 32750

Moffa, Peter E.
Sterns & Wheler
Cazenovia, NY 13035

Molzahn, Robert E.
Harza Engineering Company
150 S. Wacker Dr.
Chicago, IL 60606

Moreda, Lydia S.
City of Tampa-DPW
4604 John Moore Rd.
Brandon, FL 33511

Morey, Lawrence J.
Watson and Company
2811 E. South St.
Orlando, FL 32803

Motchkavitz, Robert J.
The Deltons Corp.
3250 SW 3rd Ave.
Miami, FL 33129

Mueller, Thomas W.
Post, Buckley, Schuh & Jernigan
5001 Lacasa Ct. #494
Tampa, FL 33617

Murali, R.S.
Dept. of Environmental Regulation
2600 Blairstone Rd.
Tallahassee, FL 32301

Musgrove, Richard J.
NW Fla. Water Mgmt. Dist.
Rt. 1, Box 3100
Havana, FL 32333

Nagy, Janet A.
Joiner Engineering, Inc.
113 W. Savidge St.
Spring Lake, MI 49456

Noss, Richard R.
M.I.T.
106 Centre St.
Milton, MA 02186

O'Toole, Michael C.
U.S. EPA
570 Frederick Lane
Hoffman Estates, IL 60195

Ott, Randy R.
Onondaga County
125 Elwood Davis Rd.
N. Syracuse, NY 13212

Perez, Armando I.
Post, Buckley, Schuh & Jernigan
7500 NW 52nd St.
Miami, FL 33166

Peterson, Ray S.
U.S. EPA
23924 Florence Acres
Monroe, WA 98727

Pew, Kenneth A.
Watermation, Inc.
8321 Celianna Dr.
Stringsville, OH 44136

Pierro, Robert H.
Civil Engineering Consultant
825 4th St. West
Palmetto, FL 33561

Pitt, Robert
Woodward-Clyde Consultants
San Francisco, CA

Porcella, Donald B.
Tetra Tech Inc.
3746 Mt. Diablo Blvd.
Lafayette, CA 94549

Poruczniak, Thomas

Pratt, Benjamin C.
Lee County-County Commissioners
P.O. Box 398
Ft. Myers, FL 33902

Ragan, Robert E.
Calocerinos & Spina
1020 Seventh North St.
Liverpool, NY 13088

Ramsey, Frederick V.
Environmental Science & Engr.
P.O. Box 13454
Gainesville, FL 32604

Ray, Larry T.

Reynolds, Tom D.
Texas A & M University
College Station, TX 77840

Risley, Clifford
U.S. EPA
536 S. Clark St., 10th Floor
Chicago, IL 60605

Saini, Mohinder K.
Dept. of the Army
MAEN-A Bldg. 677-A
West Point, NY 10996

Sartor, James D.
Woodward-Clyde Consultants
3 Embarcadero Center Suite 700
San Francisco, CA 94111

Schenk, John E.
ENCOTEC
3983 Research Park Dr.
Ann Arbor, MI 48105

Schmidt, Edward J.
University of New Hampshire
46 Madbury Rd.
Durham, NH 03824

Schwartz, Larry A.
Waccaman Reg. Plan. Council
134 Clemson Rd.
Conway, SC 29526

Sellen, Joyce
Seminole County
1 North Park Ave.
Sanford, FL 32771

Shubinsky, Robert
Water Resources Engineers
Springfield, VA

Simons, Allyson A.
Johnson Engineering Inc.
P.O. Box 1550
Ft. Myers, FL 33902

Singer, Sam N.
Ontario Ministry of the Environment
3 Massey Sq. Apt. 1601
Toronto, Ontario, Canada

Sloane, Joanna
University of Washington
2020 E. Howe St.
Seattle, WA 98112

Smedile, Joseph A.
N.E. Illinois Plng. Com.
2929 Indian Wood Rd.
Wilmette, IL 60091

Smith, John P.
Florida Institute of Technology
255 Monaco Rd., West
Melbourne, FL 32901

Smith, William W.
FMC Corp.
44 Lookover Lane
Yardley, PA 19067

Smolenski, Frank

Smolenyaic, Kevin J.
The Deltona Corp.
3250 SW 3rd Ave.
Miami, FL 33129

Snow, Madeline
Mass. Dept. of Env. Quality Engr.
88 Kenrick St.
Brighton, MA 02135

Snyder, Bruce
University of Central Florida
CEES Department
Orlando, FL 32816

Solanki, Himat T.
County of Sarasota
3012 Bucida Dr.
Sarasota, FL 33582

Sonnen, Michael B.
The M.B. Sonnen Co., Inc.
2855 Mitchell Dr. #215
Walnut Creek, CA 94598

Spence, Jeff

Stalker, Richard D.
Area Planning Board
2300 Palm Beach Lake Blvd.
West Palm Beach, FL 33409

Taylor, James S.
University of Central Florida
CEES Department
Orlando, FL 32816

Terstriep, Mike
Illinois Water Survey
Box 232
Urbana, IL 61874

Thom, Ronald M.
University of Washington
13729 27th Ave. NE
Seattle, WA 98125

Thomas, Dan

Tiemens, Myron
U.S. EPA
Washington, DC 20460

Timmons, Wilson R.
Brevard Co. Water Res. Dept.
Rt. 1, Box 195
Cocoa, FL 32922

Tindale, Steven A.
City of Tampa-DPW
Tampa Municipal Bldg.
Tampa, FL 33602

Todd, William L.
Post, Buckley, Schuh & Jernigan
6326 Presidential Ct.
Ft. Myers, FL 33907

Tolpa, Robert D.
U.S. EPA
1925 Oakton St., Apt. 2-C
Des Plaines, IL 60018

Tomasello, Richard S.
Airan Consultants
900 Overbrook Pl.
W. Palm Beach, FL 33406

Tomlinson, Richard D.
Municipality of Metro. Seattle
821 2nd Ave.
Seattle, WA 98104

Tompkins, Marian L.
Iowa Dept. Env. Quality
3211 30th St.
Des Moines, IA 50310

Tully, William P.
SUNY-CESF
109 Emann Dr.
Canillas, NY 13031

Turkeltaub, Robert
U.S. EPA
Edison, NJ 08817

Turner, Ralph R.
Union Carbide
Oak Ridge Nat'l Lab
Oak Ridge, TN 37830

Unger, Heinz K.
Reid, Crowther & Partners Ltd.
c/o R.C.P.L. #5 Oxford Park Ave.
Kingston 5, Jamaica

Valetutto, Angelo J.
Metcalf & Eddy, Inc.
524 Linden Ave.
Woodbridge, NJ 07095

VanNote, Kirby
Professional Services Group
835 No. County Rd. 18
Minneapolis, NM 55427

Wagener, Fritz
U.S. EPA
345 Courland St.
Decatur, GA 30033

Walesky, Richard
Dept. of Env. Regulation
1141 Fernlea Dr.
West Palm Beach, FL 33409

Wanielista, Martin P.
University of Central Florida
CEES Department
Orlando, FL 32816

Ward, Jeffery J.
U.S. Army Corps of Engr.
625 Shelter Creek Lane
San Bruno, CA 94066

Watkins, Frank
Dept. of Env. Regulation
9645 Bay Meadows Rd. #713
Jacksonville, FL 32216

Weldon, Ken

Welter, Gregory J.
O'Brien & Gore Engineers
1432 Lawrence St., NE
Washington, DC 20017

Wiegand, Cameron
Metro Washington Council/Gov'ts
1875 Eye St., N.W.
Washington, DC 20006

Wiemhoff, John R.
U.S. EPA
109 Norwood Ct.
Rolling Meadows, 60008

Williamson, James A.
Finkbeiner, Pettis & Strout
3709 Harley
Toledo, OH 43613

Woodruff, Gary I.
Tulsa City-Cty. Health Dept.
4616 E. 15th St.
Tulsa, OK 74112

Wycoff, Ronald L.
CH2M Hill
2521 NW 63rd Terr.
Gainesville, FL 32601

Yaap, Warren E.
Private Practice
124 Firehorn Rd.
Gulf Breeze, FL 32561

Young, Michael D.
St. Johns River W.M.D.
Rt. 2, Box 93
Melrose, FL 32666

Yousef, Yousef A.
University of Central Florida
CEES Department
Orlando, FL 32816

Zanoni, Al E.
Marquette University
1515 W. Wisconsin Ave.
Milwaukee, WI 53233